

# Monitoring of Tree Island Condition in the Southern Everglades

Report (2011-2014)

Cooperative Agreement #: W912HZ-09-2-0019



Submitted to:

Dr. Al F. Cofrancesco U.S. Army Engineer Research and Development Center (U.S. Army - ERDC) 3909 Halls Ferry Road, Vicksburg, MS 39081-6199

Jay P. Sah, Michael S. Ross, Pablo L. Ruiz, Junnio Freixa, Susana Stoffella Southeast Environmental Research Center Florida International University, Miami, FL

April 7, 2015

# **Table of Content**

Table of	Content	i			
Authors' Affiliation					
General	Background	. 3			
Executiv	ve Summary	. 4			
1 S Shark R	patio-temporal pattern in plant communities along a gradient in tree islands in iver Slough, Everglades National Park, FL	. 7			
1.1 Iı	ntroduction	. 7			
1.2 N	lethods1	10			
1.2.1	Study Area 1	10			
1.2.2	Data Collection 1	11			
1.2.3	Data Analysis1	12			
1.3 R	esults 1	15			
1.3.1	Vegetation assemblages1	15			
1.3.2	Environmental heterogeneity and species turnover1	16			
1.3.3	Vegetation change (2001/2002-2011/2012)1	17			
1.4 D	iscussion2	22			
2. D	eveloping Spectral Signature-based Tree Island Vegetation Classification	28			
2. D 2.1 In	eveloping Spectral Signature-based Tree Island Vegetation Classification	28 28			
<ol> <li>D</li> <li>D</li> <li>In</li> <li>In</li> <li>N</li> </ol>	eveloping Spectral Signature-based Tree Island Vegetation Classification	28 28 29			
2. D 2.1 L 2.2 N 2.2.1	eveloping Spectral Signature-based Tree Island Vegetation Classification	28 28 29 29			
2. D 2.1 In 2.2 N 2.2.1 2.2.2	eveloping Spectral Signature-based Tree Island Vegetation Classification	28 28 29 29 30			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         Iethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> </ul>			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3 2.2.4	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         Iethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> </ul>			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3 2.2.4 2.2.5	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         Iethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3         Temporal Change in Vegetation       3	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> </ul>			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3 2.2.4 2.2.5 2.2.6	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         lethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3         Temporal Change in Vegetation       3         Vegetation Hydrology Relationship       3	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> <li>35</li> </ul>			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3 2.2.4 2.2.5 2.2.6 2.3 R	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         lethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3         Temporal Change in Vegetation       3         Vegetation Hydrology Relationship       3         esults       3	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> <li>35</li> <li>35</li> </ul>			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3 2.2.4 2.2.5 2.2.6 2.3 R 2.4 D	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         Iethods       2         Study Area       2         Field Sampling       2         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3         Tree Island Vegetation Classification       3         Vegetation Hydrology Relationship       3         iscussion       3	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> <li>35</li> <li>35</li> <li>38</li> </ul>			
2. D 2.1 In 2.2 N 2.2.1 2.2.2 2.2.3 2.2.4 2.2.5 2.2.6 2.3 R 2.4 D Acknow	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         lethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3         Temporal Change in Vegetation       3         Vegetation Hydrology Relationship       3         iscussion       3         ledgments       4	<ul> <li>28</li> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> <li>35</li> <li>38</li> <li>42</li> </ul>			
<ul> <li>2. D</li> <li>2.1 In</li> <li>2.2 N</li> <li>2.2.1</li> <li>2.2.2</li> <li>2.2.2</li> <li>2.2.3</li> <li>2.2.4</li> <li>2.2.5</li> <li>2.2.6</li> <li>2.3 R</li> <li>2.4 D</li> <li>Acknow</li> <li>Literatu</li> </ul>	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         Iethods       2         Study Area       2         Field Sampling       3         Image Processing & Vegetation Indices       3         Tree Island Vegetation Classification       3         Temporal Change in Vegetation       3         Vegetation Hydrology Relationship       3         iscussion       3         ledgments       4	<ul> <li>28</li> <li>29</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> <li>35</li> <li>35</li> <li>38</li> <li>42</li> <li>43</li> </ul>			
<ul> <li>2. D</li> <li>2.1 In</li> <li>2.2 N</li> <li>2.2.1</li> <li>2.2.2</li> <li>2.2.3</li> <li>2.2.4</li> <li>2.2.5</li> <li>2.2.6</li> <li>2.3 R</li> <li>2.4 D</li> <li>Acknow</li> <li>Literatu</li> <li>Figures.</li> </ul>	eveloping Spectral Signature-based Tree Island Vegetation Classification       2         htroduction       2         lethods       2         Study Area       2         Field Sampling       2         Image Processing & Vegetation Indices       2         Tree Island Vegetation Classification       2         Tree Island Vegetation Classification       2         Vegetation Hydrology Relationship       2         iscussion       2         re Cited       4	<ul> <li>28</li> <li>29</li> <li>30</li> <li>32</li> <li>33</li> <li>35</li> <li>35</li> <li>38</li> <li>42</li> <li>43</li> <li>51</li> </ul>			

#### **Authors' Affiliation**

#### Michael S. Ross, Ph.D. – Associate Professor

Florida International University Southeast Environmental Research Center/Department of Earth & Environment 11200 SW 8th Street Miami, FL 33199 305.348.1420 rossm@fiu.edu

# Jay P. Sah, Ph.D. – Research Associate/Faculty

Florida International University Southeast Environmental Research Center 11200 SW 8th Street Miami, FL 33199 305.348.1658 sahj@fiu.edu

# **Pablo L. Ruiz<sup>1</sup>** – *Scientific Project Manager* Florida International University

Southeast Environmental Research Center 11200 SW 8th Street Miami, FL 33199 <u>plruiz.fiu@gmail.com</u>

#### Junnio Freixa – Lab/Field Assistant

Florida International University Southeast Environmental Research Center 11200 SW 8th Street Miami, FL 33199 jfrei006@fiu.edu

# Susana Stoffella – Research Analyst

Florida International University Southeast Environmental Research Center 11200 SW 8th Street Miami, FL 33199 stoffell@fiu.edu

<sup>&</sup>lt;sup>1</sup> Current affiliation South Florida/Caribbean Network Inventory & Monitoring Program, National Park Service.

#### **General Background**

Tree islands, an integral component of the Everglades in both the marl prairie and ridge and slough landscapes, are complex ecosystems. They are sensitive to activities associated with the Comprehensive Everglades Restoration Plan (CERP) authorized by the Water Resources Development Act (WRDA) 2000 to restore the south Florida ecosystem. More specifically, changes in hydrologic regimes associated with restoration projects are likely to affect the internal water economy of the islands, which in turn will influence tree island plant community structure and function. To strengthen our ability to assess the "performance" of tree island ecosystems and predict how these hydrologic alterations would translate into ecosystem response, an improved understanding of plant community structure and function, and their responses to major drivers and stressors is important. Built on a baseline study of vegetation structure and composition and associated biological processes over three years (1999-2002) on three tree islands in Shark River Slough (Ross and Jones 2004), a more extensive study was initiated in 2005 with initial funding from Everglades National Park and South Florida Water Management District (SFWMD). The study has been continued through 2014 with funding from US Army Corps of Engineers (USACOE). The comprehensive results of works accomplished through 2010 are described in Shamblin et al (2008) and Ruiz et al. (2011).

This report describes the dynamics of vegetation structure and composition on tree islands in the Southern Everglades, emphasizing the major works accomplished between 2011 and 2014 (Cooperative Agreement # W912HZ-09-2-0019). In 2011-2012, the field work included vegetation surveys in the hardwood hammock of eight islands, a sub-set of a 16-island network established for long-term monitoring and assessment in 2005 (Shamblin et al. 2008). The sub-set also included three Shark River Slough (SRS) tree islands, Black Hammock, Gumbo Limbo, and Satinleaf which had been intensively studied in 2000-2002 (Ross and Jones 2004). In 2011-2012, permanent plots established within three distinct vegetation zones (hardwood hammock, bayhead forest and bayhead swamp) along the N-S topographic gradient, and three transects perpendicular to the main axis on each island were re-sampled to assess the vegetation change over a ten-year period. In 2012, the scope of the project was broadened to include remote sensing work, especially to develop and assess tree island vegetation classification using spectral signature-based vegetation indices. For this reason, vegetation was sampled along N-S gradient on nine tree islands, including five within Everglades National Park (ENP), two in Water Conservation (WCA) 3A, and one in WCA-3B. The sampling sites on these islands were aligned with the pixels in the Landsat TM imagery.

The document is organized in two sections. Section 1 integrates two previous annual reports, Sah et al. (2012) and Ruiz et al. (2013), and comprehensively describes decadal change in vegetation composition along hydrologic gradients represented by transects and permanent plots in three SRS tree islands. More specifically, the section highlights the vegetation shift and successional trend that occurred in response to differences in hydrologic conditions in Shark River Slough between two decades. Section 2 explores the use of spectral vegetation indices calculated from Landsat TM imagery, and evaluates the relative accuracy of the tree island plant community classifications derived from vegetation structural data collected in the field and the vegetation indices.

#### **Executive Summary**

In the Everglades tree islands, plant community structure and composition vary on both spatial and temporal scales. In these islands, where the communities are primarily arranged along hydrologic gradients, temporal changes in flooding regime often result in a spatial shift in community composition along the gradient and determine the trajectory of community succession. However, the direction, magnitude and rate of such a change in species composition are determined by the extent of hydrologic alterations; prolonged and extreme wet events may even result in the complete loss of upland woody vegetation. In contrast, prolonged drying conditions usually set an opposite trend, i.e. the vegetation trajectory proceeds toward an expansion of woody vegetation at the expense of herbaceous plants. In the Everglades, establishment of historical hydrologic regimes is the primary goal of the ongoing restoration efforts under the Comprehensive Everglades Restoration Plan (CERP). Within the CERP, changes in water management associated with restoration will likely impact the balance and boundaries between herbaceous species and woody communities within the ridge-slough landscape, while in tree islands, the proportion of flood-tolerant and flood-intolerant woody species will change, resulting in a shift in species assemblages and tree island function. The climatological records and hydrologic data from the Shark River Slough region suggest that water level during most of the last decade of the 20th century was well above the 30-year average. In contrast, both the mean annual rainfall and water level were relatively low during the most recent decade (2001-2010). In this study, we examined the interaction between hydrology and vegetation over a 12-year period, between 1999/2000 and 2012 in three tree islands in Shark River Slough. Additionally, we evaluated twelve vegetation indices calculated using Landsat TM imagery, and assessed the accuracy of classification based on spectral vegetation indices against a classification based on vegetation structural data collected on the ground in nine tree islands. We hypothesized that relatively drier condition in recent decade would result in an increase in relative dominance of woody plants, especially flood-intolerant species, over herbaceous and flood-tolerant woody species, ultimately causing a shift in the boundaries between plant communities on the islands. Moreover, we expected that a suite of vegetation indices developed using Landsat TM spectral data would successfully distinguish among tree island community types, and thus can be useful in detecting long-term vegetation change on tree islands.

In 2011-2012, field work included vegetation sampling on three tree islands, Black Hammock, Gumbo Limbo and Satinleaf that were first studied in 2000-2002. On each of these islands, vegetation was originally sampled in two ways: (1) at 5-10 m intervals along four transects, one along the main axis, and three transects perpendicular to them, and (2) within three plots, ranging from 225 m<sup>2</sup> to 625 m<sup>2</sup> in size, selected to represent hardwood hammock, bayhead, and bayhead swamp. Along the transects, sampling protocols included an estimate of maximum height and cover class of trees and vines by species within a 2 m radius plot, and cover class of herbs and shrubs by species within a 1 m radius plot. In the permanent plots, trees and saplings of each species were censused, and DBH of each individual was measured. Additionally, the fieldwork in next two years, 2012-2014, included vegetation sampling at 309 sites along hydrologic gradients in nine islands, six within Everglades National Park (ENP), two in Water Conservation Area (WCA) 3A, and one in WCA 3B. The sampling protocols on these islands included a visual estimate of relative abundance of species in six growth forms (tree, vine, shrub, fern, forb,

and graminoid), an estimate of plant canopy height, and a measure of forest canopy cover using densiometer within nested plots at each transect point.

Species cover data were summarized using the mid-point of the cover class, and both univariate and multivariate techniques were used to examine the effects of environmental factors on vegetation structure and composition. The split moving-window (SMW) boundary analysis was used to identify boundaries between vegetation assemblages in the three tree islands. Diversity indices were calculated to examine spatial and temporal species turnover along the gradient. In addition, non-metric multidimensional scaling (NMDS) ordination was used to examine relationship between environmental factors and vegetation composition, and analysis of similarity (ANOSIM) was applied to test differences in species composition between sampling years. For assessing remote sensing and field data based classifications, the analysis included the classification of sites using user-defined dichotomous key and cluster analysis based on vegetation structure and Landsat TM spectral signature, followed by an evaluation of the different classifications using a confusion matrix.

Shark Slough tree islands revealed a more or less regular spatial pattern in plant species composition that appears to be related to topography, hydrology, and soil characteristics. On the three focal islands described in this Report, spatially differentiated vegetation occurring along the hydrologic gradient took the form of vegetation assemblages of contrasting species composition and functional representation (life forms). When plotted along the length of the transect, the boundary between adjacent vegetation assemblages varied from sharp, clearly defined peaks of compositional dissimilarity to more gradual, diffuse transition zones. However, over the last decade, the life form composition of some of these assemblages changed in response to interacting forces, including hydrology and disturbances (fire and storms). Tree cover in the hardwood hammocks decreased, especially in Black Hammock and Gumbo Limbo Hammocks, whereas the cover of graminoids, including sawgrass (Cladium mariscus ssp. *jamaicense*) increased near the boundary between marsh and bayhead or bayhead swamp forest, and within the bayhead swamps on all islands. In most of the transects and/or plots in bayhead forest and bayhead swamp portions of islands, the relative abundance of flood tolerant species like Annona glabra and Salix caroliniana declined, while that of moderately tolerant species like Chrysobalanus icaco, Ilex cassine, and Ficus aurea increased in ten years. Thus, vegetation dynamics did not always involve a simple shift in the location of fixed species assemblages, but rather the emergence of new species compositional and combinations. These results reinforce the concept that tree islands are dynamic successional communities whose expansion or contraction over time depends on the strength and duration of changes in hydrologic conditions. In general, shifts in boundaries among plant communities are presumed to initiate reductions in ecosystem resilience, resulting in regime shifts. In these three islands, however, the effects of annual variation in hydrology over the previous decade probably did not surpass the ecosystem's resilience, hence a minimal shift in boundary was observed, especially in the head, and bayhead portion of island. Furthermore, in this study, a combination of five vegetation indices calculated from Landsat imagery adequately differentiated the tree island plant communities. The five plant communities identified in this analysis differed in hydroperiod and mean annual water depth, though there was considerable overlap between communities in their hydrologic "niche". Despite these overlaps in differentiating vegetation groups, a first order map of tree island vegetation communities and estimation of temporal changes in extent seems plausible using Landsat data at broader scales and with moderate level of confidence. At the local level, however, finer scale spectral data, such as world vision 2 (WV 2) with 2 m resolution will be needed for classifying vegetation in and around tree islands, and detecting small scale changes within individual islands.

The correlation between tree island vegetation structure and hydrology suggests that hydrologic modifications brought about through the CERP will affect tree island dynamics throughout the Everglades. Depending on the magnitude of hydrologic alterations achieved by the CERP, the balance between flood-tolerant and flood-intolerant woody and herbaceous vegetation within tree islands is likely to change. Ideally, the CERP should strive to achieve system-wide hydrologic conditions that result in a spatially balanced mosaic of tree islands with different successional states, with no specific bias towards tree islands of one type or another or of a single successional state.

## 1 Spatio-temporal pattern in plant communities along a gradient in tree islands in Shark River Slough, Everglades National Park, FL

# **1.1 Introduction**

Plant communities arranged along a spatio-temporal gradient are generally viewed as a product of functional processes associated with underlying physico-chemical drivers that vary on both spatial and temporal scales. Likewise, when a community changes in species composition over time, i.e. undergoes the successional process, the conditions of drivers also are subsequently modified by the species and their interactions, generating a complex environmental gradient through time. Accompanying changes in the underlying drivers along the gradient, specific sets of the processes result in the formation of discontinuities or boundaries, representing transition zones between adjacent, distinct self-organizing structures (Allen et al. 2005). Structural and functional characteristics of such a boundary, often referred to as an 'ecotone' in the plant community literature, usually depend on whether variation in the drivers is abrupt or gradual (Wiens et al. 1985; van der Maarel 1990; Gosz 1993; Walker et al. 2003). The persistence of the position and attributes of the boundaries in space and time depends on the ability of these selfstructured identities to withstand the effects of changes in the underlying drivers on functional processes (Risser 1995; Forys and Allen 2002). In general, self-organizing structures that maintain their functional integrity, even after some changes in their elements, are usually able to persist within their prevailing spatio-temporal domain, and the boundaries between these structural systems remain intact (Forys and Allen 2002). However, when there is a significant loss of the functional processes or a change in their representation within the structural system, its boundary is likely to shift (Allen et al. 2005).

In the Everglades, tree islands are integral components of the ridge-slough and other landscapes (e.g., pine rockland, marl prairie). They provide a network of refuges for forest-dwelling plants and animals, and perform important biodiversity and nutrient cycling functions (Meshaka et al. 2002, Ross and Jones 2004; Hanan and Ross 2009; Wetzel et al. 2011). As such, tree islands serve as biological hot spots, and they may be considered as keystone habitats indicative of the overall health of the Everglades. The tree islands present within the ridge and slough landscape are complex, spatially differentiated ecosystems in their own right, often including different plant communities arranged along topographic, hydrologic and soil nutrient gradients (Armentano et al. 2002; Ross and Jones 2004; Espinar et al. 2011). In these islands, physico-chemical drivers produce a range of vegetation assemblages that vary in species composition and life-form structure, represented in the proportion of plant growth forms. Vegetation in the hardwood hammocks (HH), which lie on the most elevated portion of the islands and are rarely flooded, are mostly dominated by flood-intolerant trees, whereas the surrounding marsh has mostly floodtolerant graminoids or broad-leaved submerged, floating, and/or emergent species. Between these two extremes, the proportion of woody plants and herbaceous species varies depending on the underlying drivers of plant community composition (Sah 2004). With changes in such drivers, together with periodic disturbances (fire, hurricane), species composition may change over time, affecting the resilience of the plant communities (the capacity of a system to absorb disturbance before it shifts into a different state; Holling 1973), and ultimately the persistence of forest communities within the marsh (Figure 1.1). This study describes decadal changes in vegetation composition of Everglades tree islands, and examines whether successional processes influenced by short-term changes in hydrologic conditions have impacted the structure and

locations of boundaries between the different communities that may be distinguished in the tree island ecosystem.

Paleoecological evidence suggests that tree island nucleation, formation, and development in the Everglades began between 500 and 4,000 years before present in response to global and regional multidecadal fluxes in the periodicity and duration of flooding and drought events, which permitted the establishment and proliferation of woody vegetation in sawgrass marshes or on ridges during periods of sustained drought (Willard et al. 2002, Willard et al. 2006, Bernhardt 2011). Over time, soil accretion resulting from higher productivity rates within these incipient tree islands led to higher surface elevations and shortened hydroperiods, which in turn promoted the establishment of shrubs and trees. Several mechanisms may be responsible for increased productivity, and thus the expansion and maturation of the woody vegetation, including: translocation and accumulation of nutrients from the adjacent marsh into the incipient tree island through increased transpiration; deposition of nutrients, mainly phosphorus, in the form of bones and fecal matter of birds and mammals; and subsequent leakage of nutrients to the adjacent land in the downstream have been considered responsible (Wetzel et al. 2005; Ross et al. 2006; Givnish et al. 2008). As recently as 300 AD, many Everglades tree islands were not fully formed and exhibited many of the characteristics of a transitional community consisting of sawgrass and weedy annuals with a minor woody component (Willard et al. 2002, Stone and Chmura 2004). However, by around 1400 AD, following several extensive and prolonged local and regional drought episodes, the modern vegetation structure and hierarchy on most large tree islands in the southern Everglades had been initiated (Willard et al. 2002, Bernhardt 2011). Likewise, paleoecological studies also suggest that the location of boundaries between tree island communities and surrounding low-stature marsh vegetation might have shifted in the past, depending on hydrology, climate, or fire induced changes in surface elevation (Stone and Chmura 2004), or, since the 20<sup>th</sup> century, as a result of water management (Willard et al. 2006; Bernhardt and Willard 2009).

Hydrology is one of the major drivers of species differences along topographic gradients within individual tree islands, or among various types of tree islands in the Everglades (Armentano et al. 2002; Wetzel 2002; Ross and Jones 2004; Espinar et al. 2011). Hence, substantial changes in hydrologic conditions, whether natural or management-induced, are likely to cause quantitative and qualitative changes in plant community structure and composition, and with extreme and prolonged changes even leading to complete degradation of forest structure and extensive change in ecosystem function. Historically, such changes in hydrologic conditions were driven mainly by annual or decadal variation in the precipitation. However, during the latter part of the 19th century and continuing through the 20th century, anthropogenic alterations to the Greater Everglades impacted various elements of the landscape, including tree islands (Sklar and van der Valk 2002). For instance, management-related extreme and prolonged high water level caused loss of tree island number and coverage in Water Conservation Areas (Brandt et al. 2000; Patterson and Finck 1999, Sklar and van der Valk 2002; Hofmockel et al. 2008). In contrast, shorter hydroperiod than prevailed during the pre-drainage era resulted in the continued rapid development and succession of tree islands into well-developed forested communities in other regions (Johnson 1958, Kolipinski and Higer 1969, Willard et al. 2006). Since both adjacent tree island and marsh vegetation communities are hydrologically connected (Troxler et al. 2005; Ross et al. 2006; Saha et al. 2010; Sullivan 2011), prolonged and extreme dry or wet events may also

affect the boundary of tree islands. The climatological records and hydrologic data from the Shark River Slough (SRS) region suggest that water level during most of the last decade of the 20<sup>th</sup> century was well above the 30-year average. In contrast, both the mean annual rainfall and water level were relatively low during the most recent decade (2001-2012) (**Figure 1.2**). Such a difference in water conditions has provided an opportunity to assess the response of vegetation to the shift in hydrologic regime on three SRS tree islands that were first surveyed in 2001-2002, and then in 2011-2012.

Tree islands are also likely to be affected by the restoration efforts currently under way as part of the Comprehensive Everglades Restoration Plan (CERP) authorized by the Water Resources Development Act (WRDA) of 2000. Within the CERP, changes in water management associated with restoration will result in changes in the internal water economy of tree islands. Depending on the extent of hydrologic alterations achieved, the balance between flood-tolerant and flood-intolerant woody and herbaceous species within tree islands is expected to change, resulting in a shift in species assemblages and tree island function. Such change might result in a regime shift, i.e., large, abrupt, deleterious and long-term changes to the structure, composition, and function of a system or community (Biggs et al. 2009), and the further degradation and loss of tree islands. However, CERP activities are just as likely to lead to the formation of new tree islands in areas where they recently disappeared or, at minimum, prevent any further degradation or loss of tree islands from the Everglades.

Drying conditions usually promote the establishment and growth of woody plants in wetlands. In the Everglades, where vegetation is arranged along a hydrologic gradient from open water sloughs dominated by water lilies (Nymphaea sp.) and spikerush (Eleocharis cellulosa) to dense sawgrass (Cladium mariscus ssp. jamaicense), and finally to woody communities (Gunderson 1994; Todd et al. 2010), a decrease in water level in the landscape is expected to accelerate succession with a shift of marsh species composition toward a more sawgrass-dominated community; and the expectation for tree islands include higher overall dominance of trees over herbaceous plants, and emergence of less flood tolerant trees. This study examines the spatiotemporal variation in vegetation composition over a 11-year period between 2001 and 2012 within three SRS tree islands by; i) quantifying the species and growth form distribution along the environmental gradient, ii) assessing the response of species composition and life forms to the changes in hydrologic regime over time, iii) quantifying changes in relative importance of woody species, and iv) evaluating the effects of change in species abundance and the representation of different life-forms on the location and structure of boundaries between vegetation assemblages. We hypothesize that hydrologic differences between two census dates will result in, i) an increase in dominance of woody plants over herbaceous, ii) increase in relative abundance of flood-intolerant woody species over flood-tolerant species, and iii) concomitant change in the boundaries between different communities on the islands. Moreover, both increase in dominance of woody species over herbaceous species, and flood-intolerant species over flood-tolerant species will suggest that tree island growth, development, and succession is dependent on hydrologic fluxes, particularly during periods of prolonged droughts or below average hydroperiod.

# 1.2 Methods

# 1.2.1 Study Area

The study was conducted on three Shark River Slough (SRS) tree islands, Black Hammock (BL), Gumbo Limbo (GL) and Satinleaf (SL), within Everglades National Park (ENP) (Figure 1.3). The three islands, BL, GL and SL are situated in the eastern, central and northwestern portions of the slough, respectively. Like most large SRS tree islands, these islands are organized around slightly elevated (~1-2 m) limestone outcrops, with characteristic shape and zonation. Such islands are characterized by a well-defined 'head' associated with a topographic high or limestone outcrop that rises well above the marsh surface (Olmsted and Armentano 1997, Stone and Chmura 2004, Armentano et al. 2002, Ruiz et al. 2011) and a well-defined extended 'tail' (~1-3 km long) aligned with the prevailing surface water flow direction (Loveless 1959, Snyder et al. 1990). The 'head' of these tree islands is rarely flooded, and supports a mixture of tree species, mostly of tropical origin (Armentano et al. 2002, Ruiz et al. 2011). In contrast, a 'tail' portion of these landscapes is dominated at its upper end by a mixed-species assemblage of flood-tolerant trees, and ferns, vines and graminoids, and further downstream by tall sawgrass. These teardrop-shaped tree islands exhibit a consistent decrease in surface elevation, and canopy height from the rarely flooded heads to the seasonally flooded swamp forests and marshes in the far tail region of the islands (Olmsted and Armentano 1997; Armentano et al. 2002; Ross and Jones 2004). The subtle decrease in elevation is usually associated with decreasing productivity, lower canopy heights, and increased hydroperiod. In addition, soil characteristics vary along the gradient from head to tail. Soils in the hardwood hammock are alkaline, mineral soils with extremely high P concentrations, while soils in the seasonally-flooded tail communities are mostly organic, with decreasing P concentrations from the proximal to distal ends of woody plant-dominated vegetation (Jaychandran et al. 2004; Ross et al. 2006).

Three distinct vegetation units or assemblages grade into one another in the tail portion of SRS tree islands; Bayhead Forest (BH), Bayhead Swamp (BHS), and Sawgrass Tail (SG). These three tail units may be thought of as phases in a chronosequence, where the SG represents the earliest successional tree island community, analogous to the primordial marsh prior to tree island formation, followed by the BHS zone that represents a transitional phase between the primordial marsh and a BH climax community typifying tree island maturation in the absence of an elevated, tropical Hardwood Hammock (HH) head. In general, the SG tends to be the most dynamic and least stable of the three tree island 'tail' vegetation assemblages associated with many of the larger tree islands found within the Everglades. Sawgrass Tails are i) sensitive to hydrologic changes, which can lead to shifts in species composition and/or woody plant encroachment; ii) highly pyrogenic and susceptible to fires, which may consume all standing biomass (Wade et al. 1980), and iii) prone to episodes of decadence or die-off that lead to the complete collapse, mortality, and loss of sawgrass within this zone for several years (Wade *et al.* 1980, Alexander and Crook 1984).

The current composition and community structure is determined to a large extent by recent hydrology. The hydrologic regimes that impact the ecology of these islands are influenced primarily by annual rainfall, augmented by the southerly flow of water delivered from the Water Conservation Areas by pumps arrayed along the east-west trending Tamiami Trail (Reed and Ross 2004). Disturbances such as hurricanes and fire have also played a large role in the ecology of SRS tree islands ((Loope et al. 1994; Armentano et al. 1995, 2002; Ruiz et al. 2011).

# 1.2.2 Data Collection

# Vegetation sampling along transects

Vegetation was sampled along four transects on each of three tree islands, BL, GL and SL. One transect followed the long axis of the island, hereafter termed as 'NS transect', and the other three transects were laid in west-east direction (hereafter, WE transects), at right angles to the long transect (**Figure 2.2**). Out of three WE transects, one traversed the "head" or "hammocks", and the other two crossed the middle and lower portions of the islands; these are named as 'HH', 'BH' and 'BHS' transects, respectively, based on the vegetation present at the middle of the transect. All four transects were sampled in 2001-2002, but only the three WE transects were resampled in the spring of 2011. The length of transects and the number of sites sampled in 2001-2002 and 2011 on each transect are given in **Table 1.1**, and the coordinates in **Appendix A.1.1**.

On each transect, vegetation was sampled every 5-10 m, and the sampling protocols included, (1) an estimate of maximum height and cover class of trees and vines by species within a 2 m radius plot; and (2) an estimate of cover class of herbs and shrubs by species within a 1 m radius plot around each transect point. The cover classes used to estimate species cover in each stratum were: 1, 0-1%; 2, 1-4%; 3, 4-16%; 4, 16-33%; 5, 33-66%; and 6, >66%. Soil depth was determined by probing to bedrock with a metal rod at each surveyed location.

Island	Transect	Length of the transect (m)	# of sites sampled in 2001/2002	# of sites sampled in 2011
Black Hammock (BL)	NS	560	72	-
	WE-1	115	24	24
	WE-2	135	28	28
	WE-3	205	41	42
Gumbo Limbo (GL)	NS	1000	107	-
	WE-1	230	47	47
	WE-2	280	57	57
	WE-3	470	48	48
Satinleaf (SL)	NS	500	55	-
	WE-1	135	28	27
	WE-2	110	23	23
	WE-3	115	24	24

Table 1.1: Length of transects and number of sites sampled on each transect in three Shark Slough tree islands.

# Vegetation sampling in plots

Between 2001 and 2002, permanent plots of 25 x 25 (625 m<sup>2</sup>), 20 x 20 m (400 m<sup>2</sup>) and 15 x 15 m (225 m<sup>2</sup>) were established in the HH, BH and BHS, respectively, on each of the three study tree islands (**Figure 1.2**). Each plot was gridded into 5 x 5 m cells, whose corners and midpoint were marked by 30 cm long  $\frac{1}{2}$ " PVC stakes affixed to the ground. The center of each gridded cell was given a cell number while the cell corners were marked based on their location relative to the

exterior SW corner (0,0) of the plot. The plot and cells were set up using a compass, measuring tape, sighting pole(s), and right-angle prism. These plots were revisited, reestablished, and censused in 2012.

During the 2001/02 plot census trees ( $\geq 5$  cm) were identified to species, tagged using numbered aluminum tags, and their diameter (cm) at breast height (DBH) measured and recorded. The location of each tagged tree was recorded to the nearest 0.1 meter using the SW corner of the plot as a reference (0,0). Furthermore, if a tree had multiple stems  $\geq 5$  cm DBH, each stem was tagged with a unique ID that allowed it to be cross referenced back to its 'parent'. The DBH (cm) of these offshoot stems was measured and recorded as well. The density and species of all tree saplings (stems  $\geq 1$  and < 5.0 cm in DBH) within each 5 x 5 m cell was recorded, and assigned to one of two DBH size classes: 1 to 2.9 cm or 3 to 4.9 cm. Saplings with multiple stems between 1 and 4.9 cm DBH originating from the base of a sapling were treated as individuals but assigned or grouped to the largest sapling stem category, e.g., 1 to 2.9 cm or 3 to 4.9 cm, for that grouping. During the 2012 census, all plots were re-censused to assess changes in tree and sapling densities as well as tree ingrowth and mortality. Ingrowth of new individuals or sprouts from existing trees were tagged and added to the database.

# Hydrology

The ground surface elevation was determined at 5-10 m intervals along each transect, and all plot corners of the permanent plots by surveying via auto-level from a USGS benchmark of known elevation. We estimated hydroperiod (number of days per year of surface inundation) and annual mean water depth at each survey plot location along the transects and within the permanent plots using elevation data from topographic surveys in conjunction with long term water level records at a stage recorder situated in the open marsh at 0.5 to 1.5 km distance from the head of each island. The three stage recorders used to calculate hydrologic parameters for BL, GL and SL were P33, NP203, and G620, respectively. Daily water level at the survey sites was estimated assuming a flat water surface. Later, mean annual water depth (cm), and annual average hydroperiod (days), defined as the number of days per year that the ground surface of the plot was inundated, were calculated for each plot. These statistics were calculated over a 7year period, based on previous studies of tree island dynamics. While several authors have estimated a vegetation response time of 4-6 years in marshes (Armentano et al. 2006, Zweig and Kitchen 2009), in tree islands a strong correlation was found between variation in vegetation composition and 7-year annual average hydroperiod and water depth (Sah 2004; Espinar et al. 2011; Ruiz et al. 2011). We therefore applied a mean annual hydroperiod (days) and water level (cm) averaged over 7 contiguous water years (May 1 to April 30) prior to samplings in 2001/2002 and 2011/2012, respectively.

# 1.2.3 Data Analysis

# Transect data

Species cover data were summarized using the mid-point of the cover class, and both univariate and multivariate techniques were used to identify the vegetation assemblages along the environmental gradient, and change in vegetation structure and composition over time.

Split Moving-Window Boundary Analysis: We used a split moving-window (SMW) boundary analysis (Whittaker 1960; Ludwig and Cornelius 1987; Cornelius and Reynolds 1991) to describe variation in vegetation composition and to identify boundaries between vegetation assemblages along the surveyed transects in the tree islands. In the SMW method, the position of boundaries, defined as the location of maximum variance in species-abundance based dissimilarities between adjacent groups of sampling plots, was identified through the following steps: i) A window of even-numbered size (the number of plots) was introduced at the beginning of the transect, (ii) The window was then divided into two half-windows, iii) The cover value of each species was averaged over the plots within each half window, iv) A species abundancebased Bray-Curtis (B-C) dissimilarity was calculated between each pair of adjacent halfwindows, v) The window was then moved one plot further along the transect, repeating steps 2 and 3 until the end of the transect was reached, and vi) finally, dissimilarity profile diagrams were created by plotting dissimilarity against location of the window mid-point along the transect. In the dissimilarity profile diagram, the peaks (sharp or gradual) in dissimilarity were identified as boundaries between adjacent communities. Results of the SMW boundary analysis are scale dependent, and are affected by the choice of window size. Use of a small window size often creates noise, resulting in many peaks that represent small-scale variation in species composition. In contrast, a wide window results in fewer peaks, overshadowing the fine scale variation. First we explored the pattern using windows of different sizes (2, 4, 6, 8, 10 and 12) and from these we selected larger windows (6, 8, 10, and 12) because these resulted in boundaries which appeared to be ecologically meaningful. Multiple window sizes were selected to reduce the scale-dependency of SMW results.

We used a Monte Carlo method to test whether a boundary identified using the SMW method has a significantly higher value than expected under a null hypothesis that no distinct boundary exist between adjacent communities (Cornelius and Reynolds 1991). In the Monte Carlo method, we randomized the position of each site with its species data vector intact, and repeated the calculations of SMW dissimilarities, as outlined above, for each of the selected window sizes. We repeated the randomization 1000 times, and calculated expected mean dissimilarity and standard deviation between each pair of window-halves for a given window width. Then we calculated overall mean dissimilarity and standard deviation for each window width following Cornelius and Reynolds (1991). Since our purpose was to use multiple windows in order to reduce the scale effects, we pooled the dissimilarity value of mid-point from different window sizes. However, as dissimilarities from different window sizes are scale-dependent, we first standardized the observed dissimilarity values by calculating Z-scores for each window width. The Z-score for each mid-point for a given window-width was calculated by subtracting observed dissimilarity value from overall expected mean dissimilarity and dividing by the overall expected standard deviation (Cornelius and Reynolds, 1991). We averaged Z-scores for each site from four window sizes (6, 8, 10 and 12 sites), and plotted them against site positions along each transect. We considered the peaks that consisted of one or more contiguous sites with Zscores equal of greater than 1.65 (the value in one-tailed test: 95% confidence limit) as a significant and distinct boundary between adjacent communities (Boughton et al. 2006).

In a separate analysis, species were grouped according to their life-forms (i.e., trees, shrubs, graminoids, forbs, ferns, vines, seedlings). The mean cover of these groups at each sampling point was then used to calculate B-C dissimilarity.

*Habitat heterogeneity and species turnover:* Species turnover along the transect was represented by the B-C dissimilarity between two adjacent segments of sites in SMW boundary analysis. To examine the relationship between the degree of species turnover and variation in the environmental gradient, habitat heterogeneity was calculated as the mean absolute difference in values for elevation (and its covariates hydroperiod and water depth) and soil depth. To maintain consistency between normalized B-C dissimilarity (Z-score) and habitat heterogeneity, we first calculated absolute mean difference in the values of environmental gradient variables averaged over the sites present in each of four window sizes (6, 8, 10 and 12 sites), and then averaged the values for each mid-point for the four window sizes.

Beta diversity ( $\beta = \gamma/\alpha$ ; where  $\gamma =$  total number of species on a transect, and  $\alpha =$  mean species richness per sampling plot on the transect) was also calculated to represent overall species turnover along the gradient on each transect. To quantify overall habitat heterogeneity along the transect, we calculated the coefficient of variation (CV) for elevation, hydroperiod, water depth, and soil depth. We finally used multiple-regression to quantify the relationships between species turnover and variability in elevation, hydrologic parameters, and soil depth within and across transects.

*NMS Ordination:* Non-metric multidimensional scaling (NMDS) ordination was used to examine the relationship between species composition and environmental variables representing hydrology and soil depth. The hydrologic variables included in the analysis were 7-year annual mean hydroperiod (days) and water depth (cm). The relationship was examined using a vector-fitting procedure incorporated in the computer program DECODA (Minchin 1998). Vector fitting is a form of multiple linear regression that finds the direction along which sample coordinates have maximum correlation with the fitted variable within the ordination space. The significance of the environmental vectors was assessed using a Monte-Carlo procedure permutation test with 10,000 permutations of the species data, as samples in the given ordination space are not independent (Minchin 1998). Analysis of similarity (ANOSIM) was used to examine the differences in vegetation assemblages between two sampling years, 2001 and 2011.

# Plot data

*Tree and Sapling Dynamics:* For each plot, the total tree density was estimated by summing the total number of 'parent' trees per plot and then dividing by the total area sample per plot. Total stand basal area was calculated by adding the total basal area of all stems  $\geq 5$  cm DBH in each plot. Sapling densities and basal area were similarly calculated but all shoots were treated as individuals, for density purposes, even if they were members of a multi-stem clonal group. Finally, species' importance value (IV) was calculated by summing the relative density (R<sub>d</sub>) and relative basal area (R<sub>ba</sub>) of each species, within each plot and dividing by 2.

# **1.3 Results**

Between 1991 and 2011, daily mean marsh water elevations at the three stage recorders (P-33, NP203, & G620) used in this study closely paralleled each other within and across years, with very little variation among stage recorders (**Figure 1.4**). Annual mean water levels at the three stage recorders during both census periods, 2001/02 and 2011/2012, were at or below the 21-year average. However, during the seven-year period that preceded the 2001/02 census, annual mean water levels at stage recorders were near or above the 21-year average. In contrast, for the seven-year period that preceded the 2011/2012 census, annual mean water levels were generally at or below the 21-year average (**Figure 1.5**). The 7-year average annual mean water level prior to 2011/2012 sampling was 13.8 cm lower at G620 and 12.5 cm lower at both P33 and NP203 than prior to 2001/2002 sampling. This difference in the long-term, e.g., seven-year, annual mean water levels between censuses resulted in significantly (paired t-test; p<0.05) decreased hydroperiods across all transects and plots (**Figure 1.6**). On average, annual mean hydroperiods averaged over 7 years prior to sampling along the HH, BH and BHS transects decreased by 45, 64 and 48 days respectively, and that in those three plots decreased by 0, 94 and 70 days, respectively, between the two census dates (**Appendix A.1.2**).

# **1.3.1** Vegetation assemblages

Vegetation composition in SRS tree islands follows the topographic gradient, primarily oriented along the long axis (NS transect) parallel to the direction of the water flow, but also along the transects aligned perpendicular to the long axis. The SMW boundary analysis of the 2001-2002 species cover data along NS transects identified 2-3 significant peaks, represented by relatively high normalized B-C dissimilarity (z-scores > 1.65), resulting in 3-4 distinct vegetation assemblages, including the marsh vegetation at the far end of each transect (**Figure 1.7**). The peaks representing higher B-C dissimilarity between adjacent sample segments were identical in both species and life-form abundance data. The number and sharpness (relatively narrow and tall) of significant peaks, however, differed among islands. For instance, in GL, three significant peaks in normalized B-C dissimilarity, differentiation among four communities HH, BH, BHS and Sawgrass were clearly identifiable (**Figure 1.7**). In SL, only SMW boundary analysis based on life form abundance data revealed three significant peaks denoting the same four communities. In BL, however, the boundary separating two types of swamp forests was not distinct in either the compositional or life form analysis. Moreover, the sharpness of peaks separating adjacent vegetation assemblages was more distinct in GL than in BL and SL islands.

Plant communities identified along the NS transects were strongly associated with the hydrology gradient, as evidenced in significantly high correlation ( $\mathbf{r} = 0.81$ ;  $\mathbf{p} < 0.001$ ) between the hydroperiod vector and site positions in the ordination (**Figure 1.8**). Among the three forest communities on the islands, HH were dominated by *Bursera simaruba*, *Celtis laevigata*, *Coccoloba diversifolia*, *Eugenia axillaris*, *Ficus aurea*, and *Sideroxylon foetidissimum*. Bayhead forests, which were more diverse in total plant species composition, were comprised of a mixture of trees (*Chrysobalanus icaco, Persea borbonia, Morella cerifera*, and *Magnolia virginiana*, *Salix caroliniana*), shrubs (*Cephalanthus occidentalis*) and ferns (*Acrostichum danaeifolium*, *Blechnum serrulatum* and *Thelypteris interrupta*). BHS were comprised of one or two flood tolerant tree species (*Annona glabra, Salix caroliniana*) and a suite of graminoids and forbs. The

marsh, dominated by sawgrass (*Cladium mariscus* ssp. *jamaicense*), was present at the end of the transect. In general, tree cover decreased towards the lower end of the bayhead forests and was less than 5% in the adjacent bayhead swamp forests, where low shrubs and forbs were most abundant. Graminoid cover increased towards the tail of the island, where sawgrass constituted >80% of the total plant cover. Similar to tree cover, canopy heights in all three tree islands exhibited a strong positive association (r = 0.54, p < 0.001) with surface elevation. Hardwood hammocks that occupied the head of the island had the tallest canopies, followed by bayhead forest, and finally bayhead swamp forest (**Figure 1.7**).

The vegetation assemblages identified along the NS transects were also distinguishable on the WE transects, established in three forest zones on each island. However, the SMW analysis revealed that the boundaries between identifiable vegetation assemblages were not always distinct. The peaks representing the transition zones between adjacent communities were more distinct on HH and BH transects than on BHS transects (i.e., marsh and BHS are more similar than other adjacent pairs), and more distinct in BL and GL than SL (**Figure 1.9**). For instance, on the hammock transects, the peaks separating HH and BH were generally significant. However, unlike on the NS transect, where three forests zones were identifiable, the bayhead forests on both HH and BH transects transitioned directly into marsh. On these transects, BHS were either absent or, if present, occupied a very narrow zone that was indistinguishable in the selected window sizes in the SMW boundary analysis. A relatively narrow or absent BHS along the gradient suggests a sharp drop in tree island elevation in the direction perpendicular to the axis of the tree island.

# **1.3.2** Environmental heterogeneity and species turnover

The environmental underpinnings of the within- and among-island variability in composition and structure illustrated in **Figures 1.7 and 1.9** were sought through regression analysis with metrics of habitat heterogeneity. Calculated as the mean absolute differences in elevation, or its covariates hydroperiod and water depth, habitat heterogeneity was positively correlated with B-C dissimilarity along both NS and WE transects (**Figure 1.10; Table 1.2**). On the NS transects, many of the significant peaks (z-score > 1.65) that represented relatively high species turnover co-occurred with absolute differences in elevation of 48 cm or more, corresponding to a difference in mean annual hydroperiod of  $\geq$  185 days. Along the WE transects, however, such values were much higher in the transect through the tree island head, where the sharp decreases in elevation occurred. In contrast, in the BH and BHS zones, high species turnover could occur in association with just 15-20 cm difference in elevation, i.e. a difference in hydroperiod of <100 days (**Appendix A.1.3**).

Considering the transects as a whole,  $\beta$ -diversity differed significantly (One-way ANOVA:  $F_{2,6} = 5.7$ , p = 0.03) among the three habitat zones (HH, BH and BHS), and was higher on the HH transects than on BHS transects (**Figure 1.11**).  $\beta$ -diversity on BH transects was not significantly different from that on either HH or BHS transects. Species turnover ( $\beta$ ) in both years 2001 and 2011, was positively correlated (r = 0.83 and r = 0.84, respectively) with habitat heterogeneity, represented by CV of elevation (**Figure 1.12**). However, the relationship between species turnover and soil depth was not significant and thus, not presented here. Across all transects,  $\beta$  diversity was significantly (paired t-test: t = 7.0, p < 0.001) higher in 2011 than in 2001,

# suggesting greater microhabitat heterogeneity. The mean ( $\pm$ SE) $\beta$ values were 6.18 ( $\pm$ 0.49) and 7.56 ( $\pm$ 0.56) in 2001 and 2011, respectively.

**Table 1.2:** Pearson Correlation coefficient (r) and p-values for the relationship between mean normalized Bray-Curtis dissimilarity (Z-Score) and difference in (a) hydroperiod, and (b) water depth on nine transects, three each in Black Hammock, Gumbo Limbo and Satinleaf tree islands. HH = Hardwood Hammock, BH = Bayhead, BHS = Bayhead swamp.

			2001				2011			
Tree Island	Transect	n	Hydroj	period	Water	depth	Hydroj	period	Water	depth
			r	p-value	r	p-value	r	p-value	r	p-value
Black Hammock	HH (WE-1)	19	0.68	0.001	0.47	0.040	0.60	0.007	0.16	ns
	BH (WE-2)	23	0.79	< 0.001	0.81	< 0.001	0.82	< 0.001	0.80	< 0.001
	BHS (WE-3)	37	0.79	< 0.001	0.76	< 0.001	0.57	< 0.001	0.50	0.002
Gumbo Limbo	HH (WE-1)	42	0.47	0.002	0.37	0.014	0.51	< 0.001	0.53	< 0.001
	BH (WE-2)	52	0.27	0.050	0.43	0.001	0.23	ns	0.10	ns
	BHS (WE-3)	43	0.13	ns	0.18	ns	0.55	< 0.001	0.60	< 0.001
Satinleaf	HH (WE-1)	22	0.61	0.002	0.47	0.023	0.67	< 0.001	0.55	0.009
	BH (WE-2)	18	0.76	< 0.001	0.73	< 0.001	0.81	< 0.001	0.74	<0.001
	BHS (WE-3)	19	0.10	ns	0.13	ns	0.36	ns	0.35	ns

# 1.3.3 Vegetation change (2001/2002-2011/2012)

#### Vegetation dynamics along transects

Over the period of a decade (2001-2011), the degree of change in plant community composition varied within and among SRS tree islands. Much of the change was either near the boundary between forest and marsh communities along the HH transect, or within the BH and BHS forests on other transects. Analysis of similarity (ANOSIM) reveals that, in general, vegetation on HH transect on all three islands was not significantly different between 2001 and 2011 (**Table 1.3**). Moreover, the change in overall vegetation composition was higher in the BHS forest than in BH, and in GL and SL than in BL. The effect of hydrology on vegetation composition also varied among community types as well as among islands (**Figure 1.13**). While HH vegetation showed a drying trend in all three islands, changes in vegetation composition in BHS showed a weaker relationship to hydrology, suggesting that factors other than inter-annual hydrologic variation were also responsible for change in swamp forest composition. An obvious change was in the BHS of GL where two types of BHS forest became more distinct in 2011 than they had been in 2001 (**Figure 1.13**). The mean dissimilarity between these two assemblages was 70.6%, and the characteristic species in the eastern portion of BHS forest were sawgrass (*Cladium marsicus* ssp. *jamaicense*), willows (*Salix caroliniana*) and cattail (*Typha domingensis*), whereas

the assemblage covering the western one-third of BHS forest was primarily dominated by *Cephalanthus occidentalis* (mean cover 46%).

	Transects							
Tree island	Hardwood H	ammock	Bayhead		Bayhead swa	ump		
	R-statistic	p-value	R-statistic	p-value	R-statistic	p-value		
Black Hammock	0.011	0.272	0.071	0.024	0.131	0.001		
Gumbo Limbo	0.009	0.218	0.055	0.007	0.384	0.001		
Satinleaf	0.009	0.292	0.114	0.006	0.348	0.001		

**Table 1.3:** Global R and *p*-values from analysis of similarity (ANOSIM) testing for differences in vegetation composition between two sampling years, 2001/02 and 2011.

The temporal change in vegetation composition, particularly the change in total cover of different life-forms on nine WE transects surveyed in ten years apart, is summarized in **Table 1.4**. Along the surveyed transects in the tree islands, the change in tree cover showed mixed results. Tree cover significantly decreased in the BH transect of BL (paired t-test: t = 2.23, df = 27; p = 0.03), and in both the hammock (paired t-test: t = 4.85, df = 46; p < 0.001) and bayhead (paired t-test: t = 2.51, df = 56; p = 0.015) transects of GL. On the GL hammock transect, mean tree cover in 2011 (46.2±40.7%) declined by almost half from 2001 (79.2±68.7%). In contrast to the trend in BL and GL, tree cover in SL showed an increasing trend, though not statistically significant (**Table 1.4**). In this island, the increase in tree cover was mostly concentrated in the western half of the transects.

Tree layer vegetation on the Shark Slough islands included both flood intolerant and tolerant species. Hence, change in total tree cover along the transects was confounded by the differential response of tree species, which depended on their tolerances to flooding. Between 2001 and 2011, while the mean ( $\pm$  S.E.) cover of pond apple (*Annona glabra*), a flood tolerant species, decreased significantly from 11.1 ( $\pm$ 1.23) to 6.53 ( $\pm$ 0.78) percent (paired t-test: t = 4.3, df = 318, p<0.001), mean cover of cocoplum (*Chrysobalanus icaco*), a moderately flood tolerant species increased from 9.1% to 12.3%. Among other woody species, shrub cover increased in most transects, though the increase was statistically significant (paired t-test, p < 0.05) only in the BHS transect of BL (**Table 1.4**). Most of the increase in shrub cover in BHS of BL and GL was due to an increase in the cover of *Cephalanthus occidentalis*, whose mean ( $\pm$  S.E.) cover increased from 0.8 ( $\pm$  0.3) in 2001 to 14.7% in 2011. Moreover, cover of woody vines significantly increased on HH transects in BL and GL, and BH transect in GL.

By far the most striking change in vegetation composition in all study islands was an increase in the cover of graminoids, particularly sawgrass (*Cladium mariscus* ssp. *jamaicense*) and spikerush (*Eleocharis cellulosa*). The increase in graminoids was statistically significant (paired-t test; p-value < 0.005) in all transects except the SL hardwood hammock (**Table 1.4**). Mean graminoid cover was 1.5 to 8.0 times higher in 2011 than in 2001. Sawgrass cover increased throughout the three BHS transects. However, in the forested portion of HH and BH

transects, the increase in sawgrass cover was limited to the margins, i.e. in the BHS and marsh zones (**Figure 1.14**), suggesting that sawgrass was responding to a decrease in marsh water level in recent years in comparison to the late1990s.

	WE1 - Ha	ammock [	<b>Fransect</b>			
	Black Ha	ummock	Gumbo	Limbo	Satin	leaf
	2001	2011	2001	2011	2001	2011
Epiphyte	0.04		0.74	1.96		
Fern	5.88	5.08	15.13	12.26	3.84	7.52
Herb/Forb	4.54	9.71	5.45	5.55	4.34	6.35
Herb/Graminoid	10.50	23.54	22.82	33.34	15.54	18.98
Shrub	0.00	1.04	5.28	0.54	0.04	0.19
Seedling	6.63	10.65	18.70	13.81	3.57	6.56
Vine/Herb	0.13	6.65	2.21	0.39	6.98	3.67
Vine/Shrub	2.50	3.94	1.18	6.20	9.45	4.72
Tree	68.40	54.92	79.23	46.24	44.59	49.54
	WE2 - B	ayhead T	ransect			
Epiphyte			0.27	0.04		
Fern	9.21	9.05	26.85	24.89	10.74	15.72
Herb/Forb	3.16	7.07	10.82	13.37	3.78	3.35
Herb/Graminoid	8.30	29.63	13.42	38.55	20.59	61.41
Shrub	0.11	1.41	2.40	4.29	4.09	1.52
Seedling	2.34	3.89	11.25	2.96	0.70	3.91
Vine/Herb	1.41	13.52	6.03	0.57	0.28	0.54
Vine/Shrub	0.23	4.00	1.34	1.49	0.43	0.46
Tree	58.89	46.59	38.03	23.62	26.67	36.13
	WE3 - Bayh	ead Swan	p Transe	ct		
Epiphyte	0.02	0.06				
Fern	5.48	8.19	12.63	7.98	3.65	13.29
Herb/Forb	19.54	23.32	19.40	28.28	9.38	9.79
Herb/Graminoid	23.09	48.32	8.20	64.90	20.79	80.06
Shrub	0.82	14.99	3.65	9.96	14.50	1.25
Seedling	10.35	4.95	5.28	0.27	2.25	0.15
Vine/Herb	0.35	0.75	4.28	2.38	0.23	1.04
Vine/Shrub	0.16	0.37				1.02
Tree	21.35	19.25	7.74	11.24	6.46	12.00

**Table 1.4:** Mean cover of different life forms in plots along transects sampled in 2001 and 2011 in three tree islands. The value in bold are significantly different (Wilcoxon Matched Paired test, p < 0.05).

Besides the graminoids, several other herbaceous species with overall mean cover  $\geq 1\%$  either increased or decreased over the course of the study (**Appendix A.1.4**). Among them, the change in cover of cattail (*Typha domigensis*) was of special interest. In ten years, cattail increased in cover on the three transects where it was present in 2001, i.e., BH and BHS in BL and only BH in SL. Furthermore, it was recorded for the first time on three other transects (BHS in BL and GL, and HH transect in SL) in 2011. The increase in cattail was most evident in the BHS and marsh of GL, where its cover reached 50% at some sites (**Figure 1.15**).

## Tree and Sapling Dynamics in plots

In the HH plots, in which only tree layer was re-sampled in 2012, mean tree density and basal area, averaged over all three islands increased by 7.3% and 5.1%, respectively. However, the pattern of temporal change in both tree density and basal area differed among islands (**Figure 1.16**). While both the density and basal area in the HH plot of GL decreased during the ten year period, tree density increased by 7.3% and 23.5% in BL and SL, respectively. Moreover, the basal area decreased in BL by 6.5%, but increased in SL by 37.4%, from 36.5 m<sup>2</sup> ha<sup>-1</sup> to 50.1 m<sup>2</sup> ha<sup>-1</sup>. In both BL and SL plots, density of *Eugenia axillaris* and *Chrysobalanus icaco* increased, whereas the density of *Celtis laevigata* decreased across all islands. Especially, in GL plot, the density of *C. laeviegata* in 2011 dropped to one third of the density in 2001, from 240 stems ha<sup>-1</sup> to 80 stems ha<sup>-1</sup> (**Figure 1.16**)

Average tree densities in most of BH and BHS plots increased in ten years, between 2001/02 and 2012. In BH plots, increase in tree density ranged between 18% and 55%, and the BL-BH plot had the greatest increase in tree density (**Figure 1.17**). Moreover, while the BHS plot of GL did not have any trees during either survey, tree density increased by more than 100% in the BHS plots of BL, from no tree in 2001 to 89 stem ha<sup>-1</sup> in 2012 in SL. In contrast to an increase in woody density in tree layer, the sapling densities decreased in BH plots by an average of 64% over a decade, and the greatest decrease was in the BL Bayhead Forest plot. Nonetheless, sapling densities in BHS plots of BL and GL increased by 7% and 18%, respectively, the number of saplings decreased by 52% in the SL Bayhead Swamp plot (**Figure 1.17**).

Increase in tree and sapling density was not always paralleled with an increase in the basal area, especially in BH plots. For instance, tree basal area increased by 32% and 105% in GL and SL, respectively. But, in the BL Bayhead Forest plot, in which tree density increased by 55%, basal area decreased by 14% from 19.7 to 16.9 m<sup>2</sup> ha<sup>-1</sup>(**Figure 1.18**). In contrast, together with the tree density, tree basal area also increased in the BHS plots of both BH and SL. Sapling basal area in all three BH plots decreased over ten years by an average of 58%. The BL-BH had the sharpest decline (79%) in basal area. In contrast, sapling basal areas increased across all BHS plots by an average of 31% (**Figure 1.18**). The greatest increase occurred in the GL plot in which the sapling basal area doubled in ten years from 2.5 to 4.9 m<sup>2</sup> ha<sup>-1</sup>.

Across all BH plots, the average IV of most tree species declined between 2001/02 and 2012 (**Table 1.5**). However, the IV of three tree species *Ilex cassine, Salix caroliniana*, and *Chrysobalanus icaco* increased during this period. The most notable increase was in the IV value of the intermediately flood-intolerant *C. icaco*, whose IV increased across all BH plots and in one of the BHS plots (**Figure 1.19**). In the sapling layer, while IV of *C. icaco* remained almost unchanged in two islands, BL and GL, SL showed an increase in its importance value (**Figure 1.19**). In contrast to the increase in IV of *C. icaco*, the flood-tolerant species *A. glabra* which was dominant or co-dominant in all plots, decreased in most BH plots, but remained relatively unchanged in the three BHS plots.

Species	Bayhead Forest I	V (%)	Bayhead Swamp IV (%)		
Species	2001/02	2012	2001/02	2012	
Annona glabra	$44.0\pm13.4$	$34.3\pm4.0$	$13.8 \pm 13.8$	$50.1\pm28.7$	
Chrysobalanus icaco	$4.4 \pm 4.4$	$18.9 \pm 9.8$		$2.0 \pm 2.0$	
Ficus aurea	$11.7 \pm 11.7$	$9.5\pm9.0$			
Ilex cassine	$2.8 \pm 1.8$	$3.5\pm2.2$		$4.2\pm4.2$	
Magnolia virginiana	$14.9{\pm}9.5$	$10.6\pm5.7$		$6.7\pm6.7$	
Morella cerifera	$3.1 \pm 1.8$	$2.5\pm0.7$	$4.1 \pm 4.1$	$4.3 \pm 4.3$	
Persea borbonia	$0.7\pm0.7$				
Salix caroliniana	$17.0\pm3.6$	$20.2\pm0.2$	$15.5\pm15.5$		
Sambucus nigra ssp. canadensis	$1.5\pm0.8$	$0.4\pm0.4$	$0.0\pm0.0$		

**Table 1.5:** Mean ( $\pm$  1 S.E.) tree importance value (IV) for the three Bayhead Forest and Bayhead Swamp plots between 2001/02 and 2012.

The IV of sapling species showed much greater variability between plots and census dates than did those of the tree species. For example, in the BH plots, IV of flood-tolerant species like *A. glabra* and *Morella cerifera* decreased in 10 years (**Table 1.6**). In contrast, less dominant sapling species like *P. borbonia*, *Sambucus nigra* ssp. *canadensis*, and *Ficus aurea* increased in IV between 2001/02 and 2012. This pattern, however, was not observed in the BHS plots, in which the average IV of the three dominant sapling species, *A. glabra*, *Magnolia virginiana*, and *S. caroliniana*, remained unchanged between census dates, as did most of the other sapling species (**Table 1.6**).

**Table 1.6:** Mean ( $\pm$  1 S.E.) sapling importance value (IV) for the three Bayhead Forest and Bayhead Swamp plots between 2001/02 and 2012.

Species	BH Sapling IV (%	<b>ó</b> )	BHS Sapling IV (%)		
Species	2001/02	2012	2001/02	2012	
Annona glabra	$23.6\pm10.8$	$10.7\pm2.9$	$49.6 \pm 18.8$	$51.7 \pm 18.0$	
Chrysobalanus icaco	$37.2\pm26.2$	$50.7\pm23.7$	$0.2\pm0.2$	$0.8 \pm 0.8$	
Ficus aurea	$0.2\pm0.2$	$14.4 \pm 14.4$	$0.1 \pm 0.1$	$0.1 \pm 0.1$	
Ilex cassine	$3.3\pm2.8$	$2.6\pm2.0$		$0.5\pm0.5$	
Magnolia virginiana	$8.0\pm8.0$	$9.4\pm6.4$	$14.9 \pm 13.8$	$11.7\pm10.5$	
Morella cerifera	$21.3 \pm 12.6$	$3.1\pm1.9$	$6.1 \pm 4.2$	$7.8\pm5.0$	
Persea borbonia		$4.1 \pm 4.1$	$0.2\pm0.2$	$0.4\pm0.3$	
Salix caroliniana	$6.2\pm1.8$		$28.8\pm27.8$	$27.1\pm27.1$	
Sambucus nigra ssp. canadensis	$0.2 \pm 0.2$	$5.1\pm3.7$			

#### Vegetation change and boundaries

In general, even after ten years, boundaries between two forest types or between a forest and marsh in the head region of the islands remained distinct, suggesting that the underlying factors that define the forest zones on elevated ground elevation in these islands were resistant to small-scale annual variation in hydrology. However, the sharpness of peaks separating forest and marsh communities was more distinct in 2011 than in 2001, especially in the transition between marsh and BH or BHS forests (**Figure 1.9**). Despite a general trend in resistance by these

communities from expansion and contraction in spatial extent, there was some noticeable change in community composition at the boundaries. A few sites located at the boundaries between marsh and BH were marsh/BHS type in 2001, but had changed to BHS/BH type by 2011, indicating an increase in woody components at the boundaries (**Appendix A.1.4**). Those changes occurred mostly in the eastern portions of the GL and SL HH transects, suggesting a drying trend in the area. In contrast, the sites located at the boundary of the western portion of the two transects, BH and BHS in BL and GL were more characteristic of marsh in 2011 than in 2001, mainly resulting from an increase in sawgrass cover.

## **1.4 Discussion**

In the fixed, tear-shaped tree islands within the ridge and slough landscape of the Everglades, vegetation assemblages that differ in both species composition and functional representation (life forms) are arranged along a topographic and hydrologic gradient. These assemblages are dynamic, having changed over time in response to interacting forces, including hydrologic conditions and disturbances (hurricanes and fire). Our results show that periodic fluxes in the hydrologic regime, resulting in below average water levels and shorter hydroperiods over a period as short as one decade, promotes the growth of sawgrass in the marsh, and the establishment and growth of woody plants in the hydric woody communities of the islands. These changes in vegetation composition drive successional processes resulting in the growth, and maturation of the tree islands. However, despite these internal changes, we found minimal alteration in the position of the boundary between adjacent assemblages over the period between 2001 and 2012, with the exception of the far tail regions of the islands.

Vegetation characteristics of the three SRS tree islands are in accord with patterns described for 'fixed tree islands' present within ridge-slough landscape throughout central and southern Everglades (Loveless, 1959; and others). Four distinct vegetation assemblages - hardwood hammock, bayhead, and bayhead swamp forests, and tall sawgrass marsh were expressed most clearly along the primary axis of the islands parallel to the direction of water flow, but also were evident along secondary axes in the direction perpendicular to flow. On these secondary axes, their relative areal extent varied consistently depending on the location of the transect along the length of the island (Figure 1.9). Ultimately, vegetation zonation within tree islands is a result of water flow patterns and associated ecological processes, including biotic feedbacks that alter the local topography. In the Everglades, proposed models for the development of ridge-sloughtree island landscape have emphasized the role of water flow and the distribution of nutrients (Wetzel et al 2005; Ross et al. 2006; Bazante et al. 2006; Givnish et al. 2008, Cheng et al. 2011; Lago et al. 2011). According to these models, evapotranspiration-induced convergent flow of water is one mechanism that causes the accumulation of nutrients (mainly phosphorus) and the formation of a tree island head on topographically high ground. However, it is the strong regional water flow gradient that causes the nutrient to spread downstream in the direction of flow and to form longitudinally arranged vegetation zones (Ross et al. 2006; Givnish et al. 2008; Cheng et al. 2011). A similar process operating in directions perpendicular to flow appears to create nutrient gradients between P-rich forests on relatively high ground to P-limited marshes along the flanks of the tree islands. However, in those directions, the gradients are concentrated within a relatively short distance, resulting in narrow vegetation zones. In the tree islands we

studied, relatively narrow vegetation zones were present near the ends of the transects, especially in the HH and BH regions.

Within a tree island, boundaries between adjoining plant communities were not always distinct. While the transition between HH and BH was well-defined in the study tree islands, the transition from BH to BHS to tall sawgrass marsh was subtle, and boundaries were sometimes difficult to distinguish. While several flood tolerant tree species that occur in BH are also present in BHS, e.g., *Morella cerifera, Magnolia virginiana*, and *Salix caroliniana*, their growth is stunted in the latter. Similarly, sawgrass, whose hydrologic range is very wide, grows together with flood-tolerant tree species in BHS. Thus, a boundary between BHS with high cover of sawgrass in the understory and adjacent sawgrass marsh may not always be distinct, and changes over time depend upon the change in cover of sawgrass and other associated species. Rapid changes in sawgrass cover were largely responsible for a change in boundary characteristics along the BHS transect in GL and SL. On this transect in GL, none of the peaks were significant in 2001, whereas in 2011 three significant peaks were identified (**Figure 1.9**).

The physical factors that influence the position of boundaries among adjacent communities are likely to be the same that affect the distributions of individual species. A related concept, the spatial heterogeneity hypothesis, suggests that greater habitat (resource) heterogeneity allows the coexistence of more species (MacArthur and MacArthur 1961; Davidowitz and Rosenzweig 1998; Kumar et al. 2006). Thus, along an environment gradient, a positive relationship between habitat heterogeneity and degree of species turnover is expected. In the SRS tree islands also, we observed a positive relationship between normalized B-C dissimilarity and habitat heterogeneity in all transects (Figure 1.10), suggesting that the processes that enhance habitat heterogeneity along the gradient will result in sharp inter-community boundaries, which represent zones of high species turnover. Moreover,  $\beta$ -diversity was higher in 2011 than in 2001, suggesting that habitat resource heterogeneity also increased in the tree islands over the ten years. Generally, in periodically flooded ecosystems, such as floodplains, continuous flooding and high water level are known to form homogeneous habitat, whereas during the low water level habitat heterogeneity increases (Thomaz et al. 2007). In contrast, a fluctuating water level with periodic dry-down is likely to increase habitat heterogeneity, especially in topographically heterogeneous areas. In the Shark River Slough, annual mean precipitation and water level varied more in the last ten years than during the previous decade. Thus, we conclude that relatively dry conditions and inter-annual variability in water depth in recent decade resulted in increased β-diversity in SRS tree islands.

Tree island vegetation responds to management- and naturally-driven forces e.g., hydrology, disturbance (fire and storms), or internal ecological feedbacks. In contrast to our expectation of ubiquitous increase in woody plant cover, the change in woody cover along the transects showed mixed results. In fact, in the HH of BL and GL islands and in the BH of GL, tree cover even decreased. This finding may be the result of several interacting phenomena, including disturbances. In 2005, the study islands were hit by two hurricanes, Hurricanes Katrina and Wilma. The latter brought relatively high wind speeds (Knabb et al. 2006; Pasch et al. 2006) and caused significant tree damage on the study islands. For three years after the hurricane, cumulative tree mortality values were 17.5% and 6.2% in the HH forests of GL and BL, respectively. The high tree mortality on these islands in post-hurricane years could also be the

result of interaction of multiple disturbances, which are capable of impacting the resilience of forests, resulting in surprise outcomes (Buna and Wessman 2011). The drought that prevailed for 1-3 years (2006-2008) after Hurricane Wilma might have accentuated tree mortality on these islands. In drought years, particularly during the dry season when hammock plants use regional ground water (Saha et al. 2010), water level fell more than 70 cm below the ground surface, which might have reduced access to ground water, causing high mortality in hurricane-stressed trees.

Our results reinforce the concept that tree islands are dynamic successional communities that expand and contract over time in response to hydrologic fluxes. Between 2001/02 and 2012, BH and BHS plots saw increases in tree density and basal area as well as significant canopy development, e.g., increases in canopy volume and height. At the same time, while sapling densities decreased in the BHS plots, possibly in response to intra- and interspecific competition for nutrients and light availability resulting from canopy closure and forest maturation, the BHS plots saw both an increase in the number of trees, of which most were saplings a decade ago, and a new cohort of saplings indicating a slow but steady progression in the succession of the BHS plots into a BH forest. Temporal changes in species importance values (IV), further reinforce the concept of a successional model of tree island development over time, though a precise interpretation of changes in IV between 2001/02 and 2012 is difficult because of species-specific differences in their hydrologic tolerances and life histories over time. Once established, mature trees may persist at the extremes of their hydrologic range with no apparent deleterious effects. In general, however, flood tolerant species like *A. glabra* and *S. caroliniana* saw their IV decline while moderately flood tolerant species like *C. icaco, I. cassine*, and *F. aurea* increased.

Olmsted and Armentano (1997) postulated that a prolonged high water level during the mid-1990s followed by brief dry period was responsible for "sawgrass die-off", a pronounced, spatially extensive, and episodic decadence observed in mono-dominant stands of sawgrass in some areas of Shark Slough. In our 2001 sample, opening in the herb layer due to sawgrass dieoff was most evident in BHS transect of GL (**Figure 1.20**), where the mean cover of sawgrass was only 5.5%, one-fourth of the value observed in the equivalent transect in BL. In the Everglades, researchers have often reported sawgrass decadence, and postulated various causes, including: reduced fire frequency, nutritional imbalance, fungal infection, a boring larva (*Scirpophaga perstrialis*), and hurricane caused periphyton deposition (Hofstetter and Parson 1975; Wade et al. 1980; Alexander and Cook 1984; Olmsted and Armentano 1997; Clark et al. 2009). In the present study, we have not thoroughly investigated the cause of sawgrass die-off. However, it could have occurred due to a combination of reasons, including the extreme flooding of the mid-1990s (**Figure 1.5**).

In an area of sawgrass die-off, the succession of plants may start within months (Alexander 1967), but years may pass before full vegetation recovery is achieved. In parts of Shark Slough where open water sites due to sawgrass die-off prevailed in 2000-2001, sawgrass was still very sparse in 2007 (Ross et al. 2001; Kline et al. 2007). Wade et al. (1980) had reported that extensive areas of sawgrass decadence observed in early 1970s were not distinguishable from unaffected, healthy sawgrass stands in 1980. These studies suggest that vegetation recovery in the area of sawgrass die-off could occur within 7-10 years. In the present study as well, areas within those transects that were affected by sawgrass die-off were fully vegetated with dense tall

sawgrass ten years later. While these areas of sawgrass die-off seem to have recovered to something approaching their previous conditions, periodic sawgrass die-off events within the ridge-slough landscape have important implications. In general, sawgrass die-off on ridges negatively impacts long-term viability of the ridge-slough mosaic through shrinkage of the elevation difference between these two important features (Clark et al. 2009). Likewise, sawgrass die-off near tree islands may have effects on island development through various mechanisms. For instance, in sawgrass die-off areas, especially if that are adjacent to the wooded community, water flow velocity is usually higher than in adjacent sawgrass-vegetated area (Bazante et al. 2006). Slowdown of soil accretion process associated with greater floc transport mediated through increased flow and reduced biomass production might also exacerbate the elevation difference between the wooded portion of an island and the sawgrassdominated marsh. Such an increase in the elevation difference then could accentuate positive feedbacks for differential transpiration and P-accumulation, a mechanism suggested by researchers for the development and persistence of tree islands (D'Odorico et al. 2011). The resulting elevation difference between island and adjacent marsh then would sharpen the boundary between these components of the landscape. This may be the reason the boundaries between wooded community and sawgrass marsh in Bayhead Swamp region of GL were sharper in 2011 than in 2001. Nonetheless, in coming years the recent increases in sawgrass may disrupt such feedback, resulting in more diffuse boundaries.

In the Everglades, expansion of cattail in existing sawgrass marsh is widespread, especially in the northern and central Everglades. Cattail usually spreads into areas of prolonged hydroperiod, if nutrient conditions, especially phosphorus levels, are relatively high (Newman et al. 1998; Hagerthey et al. 2008). They largely spread into adjacent sawgrass areas by underground rhizomes, but usually do not penetrate dense vegetation until something like a peat fire or a senescent patch of sawgrass creates an opening. In this study, increase in cattail in GL was possibly associated with sawgrass die-off, which was evident as early as in late 1990s in the area adjacent to the bayhead swamp zone of the island (**Figure 1.15**).

Much of what is known about the distribution or function of tree species in Everglades tree islands, or of tree island loss in the Everglades, has been viewed in relation to hydrologic fluxes resulting in prolonged high water conditions. However, tree islands are also susceptible to the direct and indirect effects of fire, particularly during drought conditions. Fires not only kill trees but consume the rich organic soils, in the process altering water regime by lowering the surface elevation (Zaffke 1983). Under these circumstances, immediate post-fire flooding can be detrimental to tree island recovery, and may lead to their recession or complete loss (Ruiz et al. 2013). Furthermore, fire is also known to sharpen the edges of both ridges and tree islands (Givnish et al. 2008; Wetzel et al. 2008). In the last ten years, two fires, the Airboat fire of 2006 and the Mustang Corner fire of 2008 burned the marshes close to Black Hammock. These fire might have not only killed woody plants, if any were growing in marshes in reduced flow conditions, but also might have consumed the peat layer, thus causing a greater difference in elevation and sharpening the boundary between forest and marsh.

A shift in the boundary among plant communities occurs when a change in a system's ecological processes cause reduction of its resilience, resulting in regime shifts (Folke et al. 2004; Hagerthey et al. 2008). In the study islands, change in hydrology over a decade probably has not

affected each community enough to exceed its "tipping point", hence a minimal shift in boundary was observed. In a recent study, macrofossil analysis of a tree island in WCA-3A has shown that the island habitat expanded in response to the dry conditions in late 1980s, followed by a reduction during wet conditions in the 1990s (Brock et al. 2012). This implies that extreme hydrologic events are more important than average annual hydrologic conditions in shaping tree island vegetation. Moreover, in plant communities arranged along an ecological gradient, high species turnover usually occurs at the edge of discontinuities. Since there is high variability in composition within the transition zone, the turnover appears to be adaptive. Hence, communities that have low turnover at the boundary, particularly when the boundary is diffuse, may have little adaptive capacity and relatively low resilience (Allen et al. 2005). In the study islands also, species turnover was lower at the boundaries between BHS and adjacent marsh, and thus the boundaries between them would be prone to shift, especially when impacted by extreme instances of flooding or drought.

The strong correlation between tree island vegetation structure and hydrology discussed above is highly suggestive of how hydrologic modifications brought about through the CERP will affect tree island dynamics throughout the Everglades. Depending on the magnitude of hydrologic alterations achieved by the CERP, the balance between flood-tolerant and flood-intolerant woody and herbaceous vegetation within tree islands is likely to change. Biogeochemical and biogeomorphological processes, i.e., nutrient cycling and soil accretion and development, will be impacted as well. The magnitude and direction of such changes is likely to vary spatially in response to regional differences in tree island reference conditions and the extent of hydrologic modification achieved by the CERP. Thus, it is not unrealistic to think that long-term hydrologic changes brought about by the CERP could lead to the degradation and loss of tree islands from the system if the hydrologic changes that are implemented significantly alter the hydrologic regime within tree islands so much that the water regime exceeds the tolerance of the in situ vegetation. At the same time, however, tree islands currently under hydrologic stress or in a highly degradative state, e.g., ghost islands (Ewe et al. 2009), could see a recovery if hydrologic conditions conducive to woody plant establishment and growth is achieved and sustained. Ideally, the CERP should strive to achieve system wide hydrologic conditions that result in a spatially balanced mosaic of tree islands in different successional states with no specific preference to tree island type or successional state. System wide homogeneity in tree islands structure, composition, and type should be discouraged, since it is likely to lead to both the floral and faunal extirpation of rare species, which will likely result in a reduction in the overall species diversity of tree islands. At the same time, system wide homogeneity in tree islands significantly increases the chances of a system wide or regional perturbation event having catastrophic consequences.

In summary, in the Shark River Slough tree islands, spatially differentiated vegetation occurring along the hydrologic gradient consists of vegetation assemblages of contrasting species composition and functional representation (life forms). Over the last decade, cover of the flood tolerant tree species *Annona glabra* decreased, and was accompanied by an increase in the cover of the moderately flood-tolerant *Chrysobalanus icaco* between 2001/02 and 2011 along transects as well as in hydric forest plots. Furthermore, transects crossing BHS plots saw a marked increase in shrub cover, particularly in *Cephalanthus*, between 2001/02. The increase in the cover and IV of *C. icaco* along transects and in the hydric plots, respectively is an important

finding, as the paleoecological record suggests that this species is indicative of the latter phase of tree island development and maturation to BH forest. While the life form composition of some of these assemblages changed in response to interacting forces, including hydrology and disturbances (fire and storms) on only a few transects, such changes in the mixture of growth forms exactly paralleled the changes in the boundary between adjacent assemblages. Thus, vegetation changes do not always involve a simple shift in the location of fixed species assemblages, but rather the emergence of new species and structural combinations. In general, shifts in boundaries among plant communities are presumed to initiate reductions in ecosystem resilience, resulting in regime shifts. In these three islands, however, the effects of annual variation in hydrology over the previous decade probably did not surpass the ecosystem's resilience, hence a minimal shift in boundary between vegetation assemblages was observed on most transects.

# 2. Developing Spectral Signature-based Tree Island Vegetation Classification

# **2.1 Introduction**

Tree island development and maintenance is one of primary interests in restoration of the Greater Everglades. Researchers have demonstrated that over the last half century, there has been reduction in tree island number and area in both Water Conservation Areas (WCAs) and Everglades National Park (ENP), mainly due to altered hydrology and/or increased fire frequency (Patterson and Finck, 1999; Brandt et al. 2000; Sklar et al. 2013). In contrast, in some regions where hydroperiods were shorter in recent decades than during the pre-drainage era, the total area of tree islands increased (Brandt et al. 2000; Sklar et al. 2013). However, the regionally specific increases in tree island habitat were considered much less than the losses in other portions of Shark River Slough (Sklar et al. 2013). In those studies, the focus of study was the whole tree island as a unit, and the complex nature of tree islands including their constituent plant communities, and quality of tree island habitat including canopy height, tree density was virtually unaddressed. Furthermore, the transition among different communities differing in habitat quality within an island was not considered.

Within the ridge and slough landscape of the Everglades, the plant community on a large-sized tree island is characterized by a change in structure and composition with decreasing elevation from the upstream head to the downstream tail. A change in hydrologic regime, including magnitude, duration and timing of water depth and flow will not only affect the size of a tree island (Brandt et al. 2000), but may also cause a shift in relative proportion of different communities within it; in fact, internal shifts in composition may occur in the absence of changes in the absolute size of the tree island. Such a shift in community types will affect the health and performance of a tree island, as the ecosystem services provided by the tree island may considerably be altered. For instance, in the phosphorus-limited Everglades ecosystem, tree islands are considered hot spots for phosphorus accumulation (Ross and Jones 2004; Wetzel et al. 2005; Ross et al. 2006). However, phosphorus accumulation in a tree island soil is considered to be a function of hydrologic conditions, as the soil phosphorus content gradually increases from marsh to tall sawgrass, and increases dramatically within the forested communities (Ross et al. 2006). Hardwood hammocks have the highest concentrations of phosphorus, 3 to 114 times of that found in P-limited marsh (Ross and Jones 2004; Wetzel et al. 2009). Any shift in relative proportion of these communities along the gradient in an island will influenc e accumulation and spatial distribution of phosphorus, ultimately affecting the total phosphorus budget within the island. Wildlife specialized to different vegetation types on a tree island and its surroundings will also be affected. For instance, in an unimpacted area, more species of birds are found in bayhead than in willowhead and sawgrass (Gawlik and Rocque 1998), thus any shift in between communities will affect the bird populations. Thus, it is important to detect the spatio-temporal variation in plant communities in response to natural or management-induced changes in hydrologic regime, and other related drivers.

While *in situ* measurements of biophysical and structural characteristics that document vegetation type and functional health of vegetated community are important for ecosystem management, remote sensing offers a promising tool to monitor variations in such characteristics at different spatial and temporal scale. Spectral vegetation indices (VIs) that are usually dimensionless are used as surrogates for biophysical characteristics of vegetation. The indices

have been successfully used to correlate remote sensed spectral data to plant community biophysical parameters such as biomass, leaf area index, percent green cover, and transpiration (Broge and Leblanc 2000; Haboudane et al. 2004, Weiss et al. 2004, Nagler et al. 2005, Vescovo & Gianelle 2008). T ree island plant communities arranged along hydrologic gradients differ in canopy cover, canopy height, and relative cover of different life forms, including trees, vines, shrubs, ferns, forbs, and graminoids. These differences in structure and composition of vegetation are likely to affect the spectral signature recorded by the sensors, and thus the different communities will have different signatures, making the indices more useful for tree island study.

This study combines field compositional and structural plant data along with Landsat TM imagery to: 1) characterize tree island vegetation communities; 2) develop a relationship between tree island community types and remote sensed spectral indices; and 3) track long term, multi-decadal, changes in tree island vegetation using spectral indices as biophysical indicators of community type and structure.

# 2.2 Methods

# 2.2.1 Study Area

The study was conducted on nine large fixed<sup>2</sup> tree islands located in the Everglades National Park (ENP) and Water Conservation Areas (WCAs) 3A and 3B. Six tree islands are in the ENP, two in WCA 3A, and one in WCA 3B (**Figure 2.1**). In ENP, four tree islands, Black Hammock, Gumbo Limbo, Johnny Buck and Satinleaf are in the Central Shark Slough (CSS), and two islands, Chekika and Heartleaf, are in the Northeast Shark Slough (NESS). Chekika is within the Blue Shanty flow-way, 1-mile south of the planned 2.6-mile bridge along the Tamiami Trail, and Heartleaf is within 0.7 mile to the south of the newly constructed one-mile Bridge (Figure 2.1). Among the three islands in the conservation areas, WCA3A-266 is in the southern part of WCA3A, where the water level is relatively high due to impoundment associated with the Tamiami Trail, TI-66 is in the central WCA-3A, and the third island, WCA3B-12 is in the central part of the WCA3B. While all tree islands in as well as in both water conservation areas will be impacted by the hydrologic changes resulted from the ongoing water managements and future project activities under Central Everglades Planning Project (CEPP), Chekika and Hearteaf islands located immediate downstream from the Tamiami Trail bridges are likely to be directly impacted by increased water flow from WCA into the ENP (USACOE 2014).

All nine tree islands have a topographic high or limestone outcrop, at the "head", which rises well above the marsh surface, and a well-defined characteristic vegetated "tail" paralleling the regional surface water flow direction and extending several hundred meters from the "head" (Armentano et al. 2002). The "head" portion of all the study tree islands, except for WCA3B-12, consists of a closed-canopy, intact or nearly intact, tropical hardwood hammock typified by a species assemblage of tropical and temperate hardwood trees (e.g., Gumbo Limbo (*Bursera simaruba*), Southern Hackberry (*Celtis laevigata*), Satinleaf (*Chrysophyllum oliviforme*), Pigeon

 $<sup>^{2}</sup>$  The term fixed tree islands is used by some to describe tree islands that have a characteristic topographic bedrock outcrop located at the top or "head" of a tree island.

plum (*Coccoloba diversifolia*), White stopper (*Eugenia axillaris*), Strangler fig (*Ficus aurea*), Wild mastic (*Sideroxylon foetidissimum*), and Paradise tree (*Simaruba glauca*)). At WCA3B-12, the exception, the upland "head" portion had very little tree cover and showed signs of recent anthropogenic activities. All nine tree islands, however, exhibited the distinctive downstream woody vegetative swamp forest "tail" terminating in a sawgrass marsh community. In all of the study tree islands, the "tail" environment was characterized by the presence of flood-tolerant trees (e.g., Pondapple (*Annona glabra*), Coco-plum (*Chrysobalanus icaco*), Dahoon holly (*Ilex cassine*), Sweet bay (*Magnolia virginiana*), Wax Myrtle (*Morella cerifera*), Red bay (*Persea borbonia*), and Willow (*Salix caroliniana*), ferns (e.g., *Acrostichum danaeifolium, Blechnum serrulatum, Osmunda regalis, Thelypteris interrupta*, and *Thelypteris palustris* var. *pubescens*), vines, forbs, and graminoids.

Fixed tree islands within the Greater Everglades (GE) have had a long history of anthropogenic habitation and use. Archeological finds and historical accounts depict a long history of indigenous people, as well as Europeans visiting and living on the "heads" of most, if not all, fixed tree islands within the Everglades (Willoughby 1898; Carr 2002). Furthermore, historical aerial photography dating back to the late 1930s and early 1940s show the "heads" of many fixed tree islands as either being devoid of trees or in a post-abandonment recovery phase. With this history of human use, they may be considered "second-growth" forests.

# 2.2.2 Field Sampling

# Vegetation data

Tree island and marsh vegetation data were collected along the longest axis of each tree island. Plots were spaced at approximately 30 to 42.2 meter intervals coinciding with the centroid coordinate of Landsat TM 30 x 30 meter pixels (**Appendix A. 2.1** – tree island plot coordinates). Sampling plots for all tree islands originated in the marsh just north of the island "head" and continued through the swamp forest "tail" and into the terminal downstream sawgrass marsh (**Figure 2.1**). A total of 309 plots were sampled between October 2012 and February 2014 (**Table 2.1**).

Table 2.1: Tree islands, their locations (NAD 1983 UTM R17) and the number of plots sampled

Tree Island	Region	Easting	Northing	Number of Plots	Dated Sampled
Black Hammock	ENP	531300	2832630	18	10/18/2012
Gumbo Limbo Hammock	ENP	526020	2834820	42	10/12/2012
Heartleaf Hammock	ENP	547620	2848170	29	1/18/2013
PSU 66 TI	WCA3A	523710	2867430	50	2/11/2013
Satinleaf Hammock	ENP	524490	2838030	20	11/1/2012
WCA3B-12	WCA3B	546300	2857380	49	12/7/2012
Chekika Island	ENP	534360	2847510	40	11/14/2013
Johnny Buck	ENP	528270	2834700	41	11/27/2013
WCA3A-266	WCA3A	518070	2853150	20	2/28/2014

Using a nested plot design, the presence of plant species present within a plot was recorded. In each plot, relative abundance of species in each of six growth form types: tree, vine, shrub, fern, forb, and graminoid, and the total cover of each growth form were estimated. The overstory trees, defined here as woody individuals with a height above two meters, and woody vines were sampled within a 2.5 m radius plot and ranked using an ordinal scale where species with the greatest abundance were assigned a "1". Shrub, fern, forb, and graminoid species were similarly sampled and ranked within a 1 m radius plot centered within the 2.5 m plot. Later, these ranks were used to estimate the abundance of each species. In three tree islands (Chekika, Johnny Buck, and WCA3A266), however, absolute cover of each species in different lifeform groups was estimated *in situ* using a modified Braun-Blanquet scale based on the following six cover categories: 1: <1%; 2: 1-4%; 3: 4-16%; 4: 16-32%; 5: 32-66%; & 6: >66%.

Plant canopy height (m) within each 2.5 m plot was estimated and categorized into one of nine height categories: Cat 1: 0, 2: 0-1, 3: 1-2, 4: 2-3, 5: 3-5, 6: 5-7, 7: 7-10, 8: 10-15, and 9: >15. At each plot center, mean forest overstory density (canopy cover) was estimated by taking four densiometer readings facing in each of the four cardinal directions (i.e., North, East, South, and West) (Lemmon 1956). The densiometer had a convex mirror, engraved with 24 squares in it. It was held at breast height, and canopy closure was estimated by calculating the number of squares (or quarters of each square) covered by the image of the canopy. The total number was then multiplied by 1.04, and averaged over the four readings to calculate percent canopy cover in each plot.

# Hydrology data

Within each vegetation plot, three representative water depth measurements were taken by measuring the distance between the ground surface and the water table surface above the ground. At the few plots where the water table was below the ground surface, a small 3-cm radius hole was dug and allowed to equilibrate while the vegetation sampling within the plot was completed. The water table elevation, at these plots, was then estimated by measuring down from the soil surface to the top of the water table. These measurements were recorded as negative values to indicate that the water table was below the ground surface. No water depth measurements were taken at the "head" of the study tree islands because of the difficulty to dig through the bedrock. Water depths at the "head" of each tree island were recorded as zero.

In conjunction with EDEN (Everglades Depth Estimation Network, http://sofia.usgs.gov/eden) water surface elevation data, field water depth measurements were used to estimate the hydrologic conditions at each sampling site. EDEN acquires water level data from a network of stage recorders throughout the Everglades, and produces interpolated daily water surface estimates (Palaseanu and Pearlstine 2008). Tree island plot ground elevation was estimated by subtracting the mean water depth from the EDEN water surface elevation for the marsh adjacent to each tree island on the day it was sampled. For the plots on the head of seven tree islands within ENP and WCA3B, the ground elevation was based on the relative elevation survey from the water edge in the marsh to five locations in the hammocks (Ross and Sah 2011). Likewise, the ground elevation data for tree island hammock plots in WCA3A were obtained from Furdi and Volin (2007). Mean annual water depth, and discontinuous hydroperiod (i.e., the number of days per year when the location had water depth > 0 cm for each plot) were then estimated based

on the calculated plot ground elevation and a time series data (2000-2013) of water surface elevation available from the EDEN database. Previous studies have found that tree island vegetation composition are well correlated with the previous ~7 years of hydrologic conditions (Ross and Jones 2004; Sah 2004, 2012; Ruiz et al. 2013). Thus, we averaged hydroperiod and mean annual water depth for the seven water years (May 1st – April 30th) prior to each sampling event to examine the relationships between hydrologic parameters and tree island plant community types.

# 2.2.3 Image Processing & Vegetation Indices

# Atmospheric Correction & Image Rectification

Three cloud and haze free Landsat TM images with near identical solar elevation angles (2 November 1985, 6 November 1998, and 10 November 2011) spanning a 26-year period between 1985 and 2011 were selected for this study. The 10 November 2011 imagery was the only Landsat TM image available that best matched our field sampling season for this project (Table 2.1).

All three Landsat TM imagery were atmospherically corrected in ERDAS Imagine using the image-based atmospheric transmittance (COST) model developed by Chavez (1996)<sup>3</sup>. The COST model, which takes into account the multiplicity effect of atmospheric transmittance, is an improvement on the dark objection subtraction (DOS) model (Chavez 1996). The COST model, however, has been found to underperform under high relative humidity (Wu et al. 2005). Thus, to further reduce any variance in pixel reflectance between the images resulting from non-surface factors, the 1985 and 1998 images were rectified to the 2011 image (Wilson and Sader 2002, Jensen et al. 1995). Image rectification was achieved by regressing the spectral reflectance of pseudoinvariant bright (sand, concrete) and dark (water) pixels from each band in the target image (1985 and 1998) to the corresponding pixel in each band of the reference image (2011).

# Vegetation Indices (Evaluation)

A spectral vegetation index, a quantitative measurement that indicates vegetation characteristics, is generated using mathematical combinations of reflectance values from different spectral bands. Vegetation indices (VIs) are largely used to enhance the vegetation signal in remote sensing data (Jensen 2005). The VIs that extract the unique spectral signature of green vegetation are useful for differentiating vegetation from the spectral signatures of other earth materials, discriminating among vegetation. More than two dozens of VIs are available for studying various aspects of vegetation (Bannari et al. 1995; Jensen 2005). The indices that were evaluated for their ability to accurately classify five vegetation communities within tree islands in the Florida Everglades are listed in **Table 2.2**. In order to assess the ability of each index to detect and classify the vegetation communities, we extracted the VI values for each vegetation plot on nine tree islands from the 2011 Landsat imagery. We then calculated the mean ( $\pm$  SE) VI values averaged over all tree island plots, and plotted them against the target vegetation types identified

<sup>&</sup>lt;sup>3</sup> An ERDAS Imagine model is available from the RS/GIS Laboratory at Utah State University (<u>http://earth.gis.usu.edu/imagestd/</u>).

using the dichotomous key. The indices were evaluated based on their ability to differentiate between the five classes. If spectral signatures overlapped, the indices were considered to be unable to accurately distinguish between communities. From the VIs considered good enough to differentiate each vegetation class, we plotted those indices against each individual island in order to see which index performs better on an island by island basis. This allowed us to narrow down the best spectral vegetation indices to use in classifying vegetation based on spectral signatures. Once a group of VIs was selected, we then calculated the values of the selected indices for each vegetation plot also from 1985 and 1998 Landsat imagery of three tree islands (Black Hammock, Gumbo Limbo and Satinleaf), for which ground vegetation data from multiple surveys were available.

Vegetation Index	Formula	Reference
<b>DVI</b> : Difference	$\mathbf{D}\mathbf{W} = \mathbf{P}_{\mathbf{W}} = \mathbf{P}_{\mathbf{D}}$	Broge and Leblanc 2000
Vegetation Index	$DVI - R_{Nir} - R_{Red}$	Broge and Lebrane 2000
EVI: Enhanced	$EVI = 2.6(R_{Nir} - R_{red}) / (R_{Nir} + 6(R_{Red}) + 7.5(R_{Blue})$	Nagler et al. 2005
Vegetation Index	+ 1.0)	Rugier et al. 2005
GRVI: Green-Red	$GRVI = R_{CHVI} - R_{RVI} / R_{CHVI} + R_{RVI}$	Motohka et al. 2010
Vegetation Index	Green Reed Reed Reed	1010tolliku <i>el ul</i> . 2010
<b>IPVI</b> : Infrared Percentage	$IPVI = R_{Nir} / (R_{Nir} + R_{Pod})$	Crippen 1990
Vegetation Index	$\mathbf{H} \mathbf{V} \mathbf{I} = \mathbf{H}_{\mathrm{MIF}} \mathbf{I} \mathbf{H}_{\mathrm{Red}}$	chippen 1990
MCARI1: Modified		
Chlorophyll Absorption	$MCARI1 = 1.2[2.5(R_{Nir} - R_{Red}) - 1.3(R_{Nir} - R_{Green})]$	Haboudane et al. 2004
Ratio Index		
MSR: Modified Simple	$MSR = ((R_{22}, /R_{22}, -1))/\sqrt{(R_{22}, /R_{22}, +1)}$	Haboudane <i>et al.</i> 2004
Ratio	More ((RNir/Red 1))/ (RNir/Red 1)	11400444110 07 W. 2001
<b>NDVI</b> : Normalized		
Difference Vegetation	$NDVI = R_{Nir} - R_{Red} / R_{Nir} + R_{Red}$	Rouse <i>et al.</i> 1973
Index		
<b>PPR</b> : Plant Pigment Ratio	$PPR = R_{Green} / R_{Blue}$	Warren & Metternicht 2005
<b>PVR</b> : Photosynthetic	$PVR = R_{Cross} / R_{Pod}$	Warren & Metternicht 2005
Vigor Ratio	r vit – rtgreen / rtked	Warten & Wetterment 2005
<b>RDVI</b> : Renormalized		
Difference Vegetation	$RDVI = (R_{Nir} - R_{Red})/\sqrt{(R_{Nir} + R_{red})}$	Haboudane et al. 2004
Index		
RDVI-LAI:		
Renormalized Difference	$RDVI_{-}I \Delta I = 0.0918 evn(6.0002 * RDVI)$	Haboudane <i>et al</i> 2004
Vegetation Index - Leaf	$RDVI^{-}LIII = 0.0710 \exp(0.0002 - RDVI)$	Haboudane et ul. 2004
Area Index		
SLAVI: Specific Leaf	$SLAVI = B_{Nin} / (B_{Pod} + B_{Nin})$	Lymburner <i>et al</i> 2000
Area Vegetation Index	SLITVI - RNIF (RKed + RMIF)	Lymound et ul. 2000

Table 2.2: List of vegetation indices (VIs) used in this study

# 2.2.4 Tree Island Vegetation Classification

#### Dichotomous Key

Tree island vegetation composition and structure, including canopy height and relative cover of different life forms, such as trees, shrubs, vines, ferns and graminoids, vary along hydrological gradients (Sah 2004). Based on hydrologic position, vegetation height and relative cover of different life forms along the gradient, a dichotomous key was developed to classify each

vegetation plot sampled (Table 2.3). Classification of woody vegetation followed the nomenclature used in Armentano et al. (2002), whereas the herbaceous vegetation was distinguished mainly into two types: 'sawgrass' and 'marsh'. All tree islands that we sampled in this study had a topographic high or limestone outcrop, at the "head", mostly with tropical hardwood species, followed by a gradual decrease in woody cover and vegetation height towards the "tail" of the island. The vegetation in the 'far tail' region of tree islands was primarily dominated by sawgrass and/or other herbaceous species. Some plots, for which the structural data were missing, were not classified using the dichotomous key. That constituted only 4% of the data.

**Table 2.3:** Dichotomous key for the classification of vegetation plots within tree islands in Shark River Slough,

 Florida.

1a. Trees or shrubs on a limestone outcrop or topographic high:	
	Iardwood Hammock.
1b. Trees, shrubs, or graminoids not on a limestone outcrop or topograp	hic high: go to <b>2</b> .
2a. Tree and shrub cover >10%:	go to <b>3</b> .
2b. Tree and shrub cover < 10%:	go to <b>6</b> .
3a Mean vegetation height $> 1$ m	go to A
Sa. Wean vegetation height $\leq 4$ m.	
30. Mean vegetation height < 4 m	go to 5
4a. Mean vegetation height $\geq$ 4 m and tree cover $\geq$ 40%:	Bavhead Forest.
4b. Mean vegetation height $\geq$ 4 m and tree cover $<$ 40%:	Bayhead Swamp.
5a. Mean vegetation height $< 4$ m, tree cover $< 40\%$ and shrub cover be	tween 10-100%
	Bavhead Swamp.
5b. Mean vegetation height $< 4$ , tree and shrub cover $< 10\%$	
	-
6a. Mean vegetation height between 1.5 and $\leq$ 3.5 m, tree & shrub cover	$r \le 10\%$ : and sawgrass cover $\ge$
Sawgra	ss Marsh.
4b. Mean vegetation height $< 1.5$ meters, tree and shrub cover $< 1\%$ and	l sawgrass cover
< 50%:	Marsh.

# Structural Classification

Vegetation plots were classified using the structural data collected in the field. Five structural parameters were used; mean canopy cover, plant canopy height, total woody cover (shrubs and trees), graminoid cover, and dryness (wet vs dry). A 'dryness' value of 100 was assigned to the plot that had dry conditions year round, and all other sites where there was standing water at the time of sampling were given a value of 0. This was based on the prior findings that have shown that the head of these islands is rarely flooded, whereas the vegetation communities at low elevation is flooded for varying period within a year depending on their location along the elevation gradient. Cluster analysis was carried out using Euclidean dissimilarity as the distance measure coupled with Ward's linkage method. In some plots, one or two structural parameters were missing, and thus were not included into cluster analysis. Those plots were located on three tree islands (Gumbo Limbo (1), TI-66 plots (11), and WCA3B-12(14)), and they constituted 8% of the total plots. The vegetation plots were classified into a maximum of five classes based on the resulting cluster groupings of the analysis.

50%)

# Spectral Signature Classification

Vegetation plots in all three year's images (1985, 1998 and 2011) were classified based on the spectral signatures of best spectral vegetation indices (VIs) which in turn were determined following the procedure described in 2.3.2. Cluster analysis was performed using Euclidian distance as the similarity measurement coupled with Ward's linkage method. The selected vegetation indices were used to calculate Euclidean distance. Each plot was initially labeled with the dichotomous key classes which were helpful in visually identifying the clusters. Vegetation plots were classified into a maximum of five classes based on the number cluster groupings chosen a priori.

# 2.2.5 Temporal Change in Vegetation

Vegetation classification results obtained using the three different methods described above were cross-tabulated against each other, and percent agreement between classifications was calculated. Finally, a temporal change in vegetation types over 26 years, at 13 year intervals, were examined by comparing the same sites classified using selected vegetation indices calculated from 1985, 1998 and 2011. The change in vegetation types were assessed only for three SRS islands (Black Hammock, Gumbo Limbo and Satinleaf).

# 2.2.6 Vegetation Hydrology Relationship

Mean annual water depth and hydroperiod were calculated for each plot, and then summarized by five community types (hardwood hammock, bayhead, bayhead swamp, sawgrass and marsh) identified using the dichotomous key described above. Discrimination among tree island plant communities along the hydrologic gradient across all islands was assessed using skewed normal distribution pattern of these communities against the hydrologic parameters. Skewed normal probability distribution is an extension of the normal distribution that allows for non-zero skewness. It is used in the Everglades Vegetation Succession Model (ELVeS) and has been demonstrated to be successful in discriminating the herbaceous community types in the Everglades (Pearlstine et al. 2011). We used the utility software, ELVeSkew ver. 1, developed by Ecological Modeling Team of South Florida Natural Resources Center (SFNRC), Everglades National Park, to get a best estimate of four parameters of skewed normal distribution from the frequency histogram values for the five community types.

# 2.3 Results

# **Evaluation of Spectral Vegetation Indices**

The use of vegetation indices, derived from Landsat imagery in classifying tree island plant communities revealed mixed results, since not all spectral vegetation indices proved adequate in discriminating among the target tree island plant communities. Among the twelve VIs evaluated for differentiating the target communities, five indices were better suited than others. Those five indices were EVI, MSR, RDVI, RDVI-LAI, and SLAVI, and the mean values of these indices
significantly differed (One-way ANOVA; p <0.05) among target vegetation types (**Figure 2.2**). Except for one or two indices and in few individual islands, the results were consistent whether data analyzed were from all islands together or for individual islands. In the case of all islands together, the most difficult to differentiate were hardwood hammock and bayhead, for which the mean values of EVI, MSR, and RDVI did not significantly differ (**Figure 2.2**). Likewise, when analyzed for individual islands, one or more of these indices were not adequately differentiating between bayhead swamp and sawgrass in Satinleaf and Johnny Buck, bayhead and bayhead swamp in Chekika and WCA3B-12, and hardwood hammock and bayhead in Chekika and Johnny Buck (**Appendix A.2.2**).

Four indices, DVI, IPVI, MCARI1 and NDVI, were good enough to differentiate five target vegetation types in all islands together, except for a considerable overlap between hardwood hammock and bayhead (**Figure 2.2**). Nonetheless, when the analysis was extended to the individual islands, these four indices proved to be inadequate in differentiating the five communities, and there was significant overlap among communities with respect to the indices (**Appendix A.2.2**). Similar results were observed when the values for another three indices, GRVI, PPR, and PVR) were examined. While GRVI and PVR failed to differentiate among three wet communities, bayhead swamp, sawgrass and marsh, and also between hardwood hammock and bayhead, PPR proved to have lowest power in discriminating the target communities, as mean values for most of the pairwise comparisons between communities were not significantly different (One-way ANOVA; Bonferroni test: p > 0.05). These three indices also performed very poorly when the data were analyzed for individual islands (**Appendix A.2.2**).

# Classification Comparison

The vegetation in tree islands and adjacent plots was broadly classified in five categories, hardwood hammock, bayhead, bayhead swamps, sawgrass and marsh. The utility of structural parameters and spectral vegetation indices (VIs) in differentiating those communities were examined by comparing the structural and spectral classification results against the community classification carried out using expert-developed classification key. The classifications developed at both scales, nine islands together and individual islands, were compared.

Plant communities along the hydrologic gradient on tree islands evidently varied in mean canopy cover, plant canopy height, total woody cover (shrubs and trees) and graminoid cover, resulting in structural parameter-based cluster analysis (hereafter 'structural classification') results very similar (percent agreement = 85%) to the classification achieved via the expert-developed classification key (**Table 2.4**). This result suggests that the structural parameters considered in cluster analysis were adequate in differentiating the target communities. In contrast, the VIs-based classification across all islands had slightly lower agreement with the dichotomous key and structural classification, and between structural and VI-based classification were 79.4% and 78.0%, respectively. The errors in agreement between VI-based classification and others mostly centered on hardwood hammocks, which were not well differentiated in VI-based classification (**Figures 2.3-2.5**).

	Black Hammock	Gumbo Limbo	Johnny Buck	Satinleaf Hammock	Chekika Island	Heartleaf Hammock	WCA3A- 266	TI-66	WCA3B- 12	All Islands
Dichotomous vs. Structural	83.33	90.24	82.93	80.00	87.50	82.14	85.00	74.36	94.29	84.75
Dichotomous vs. Spectral	83.33	80.49	78.05	80.00	77.50	92.86	90.00	58.97	85.71	79.43
Structural vs. Spectral	88.89	82.93	70.73	90.00	70.00	82.14	85.00	69.23	80.00	78.01

 Table 2.4: Summary of classification comparison results (percent agreement)

The agreement between the VI-based classification and the other two classifications was not consistent throughout the study area. The agreement among the classifications was mostly above 80% for the islands within the ENP and WCA3B (Figures 2.3-2.5, Appendix A.2.3). Only in two islands, Chekika and Johnny Buck, were agreements among classifications lower, between 70% and 80% (Appendix A.2.3). However, the tree island (Ti-66) in the southern WCA3A, where the water level remains relatively high throughout the year, showed great discrepancy between VI-based classification and the others. For TI-66, the agreement of VI-based classification with dichotomous key and structural classification was only 59% and 69%, respectively (Appendix A.2.3).

# Vegetation change (1985-2011)

The vegetation types identified in the plots using VI-based classification for 1985, 1998 and 2011 were almost identical in two islands, GL and SL (**Figure 2.6**). In BL, there were fewer sites classified as marsh or sawgrass in 1985 and 1998 than in 2011, and no plot was classified as bayhead swamp in the earlier two years. In all three islands, however, HH was not identified in VI-based classification of sites in 1985, 1998, or 2011. Moreover, over the period of analysis (1985-2011), these islands showed slight changes in vegetation types along their major axis, i.e. in the direction of water flow. In GL and SL, where the changes were minimal, few sites changed from bayhead swamp or marsh to sawgrass type at the head end of the islands. In BL, however several sites changed from sawgrass to marsh type in the tail portion of the island. In addition, 30-40% of sites that were identified as bayhead in 1985 and 1998 changed to the bayhead swamp type in 2011.

# Vegetation:hydrology relationships

The vegetation communities differed significantly in hydroperiod and mean annual water depth (**Table 2.5**), though the skewed normal probability distribution (SNPD) results showed that there was some overlap among communities along the hydrologic gradient (**Figure 2.7**). The SNPD parameters for five communities for both hydroperiod and mean annual water depth are given in **Table 2.6**. The four parameters are location, scale, shape and max. Location signifies the shift of the distribution from the horizontal origin, scale is the statistical dispersion of the distribution, and shape determines the skewness of the distribution. Max is simply the maximum value of the distribution, and is used to standardize height of the distribution curve between 0 (community type not found) and 1 (community type most frequently found). Most of overlaps along hydrologic gradient were between bayhead swamp, sawgrass and marsh communities.

Hydrologic metrics	Community type	Ν	Median	Mean	SD	Minimum	Maximum
	НН	8	0.0	3.3 <sup>a</sup>	9.2	0.0	26.0
	BH	91	194.0	187.2 <sup>b</sup>	50.1	53.0	307.0
Hydroperiod (Days)	BHS	67	240.0	235.0 <sup>c</sup>	58.1	99.0	321.0
(Days)	S	23	278.0	284.4 <sup>d</sup>	30.4	238.0	349.0
	М	9	321.0	317.1 <sup>d</sup>	20.4	282.0	341.0
	HH	8	-66.3	-68.3 <sup>a</sup>	19.8	-92.2	-31.6
Annual Mean	BH	91	-1.0	-0.8 <sup>b</sup>	6.5	-20.4	17.8
Water Depth	BHS	67	6.5	6.6 <sup>c</sup>	9.3	-13.0	28.3
( <b>cm</b> )	S	23	13.2	15.6 <sup>d</sup>	9.0	3.8	39.6
	Μ	9	22.0	22.2 <sup>d</sup>	6.4	12.6	31.3

**Table 2.5:** Two hydrologic metrics (hydroperiod and annual mean water depth) values for five community types.

 Different letters in the superscript (mean values) indicate significant difference between community types.

 Table 2.6: Parameter estimates of skewed normal distribution from the frequency histogram values for the tree island community types

Hydrologic metrics	Community type	Location	Scale	Shape	Max
	HH	19.995	360	-165.00	0.798
	BH	239.995	80	-4.99	0.721
Hydroperiod (Davs)	BHS	309.995	80	-4.99	0.700
(2435)	SG	239.995	50	5.01	0.666
	М	349.995	40	-370.00	0.704
	HH	-54.995	30	-16.00	0.755
Annual Mean	BH	15.005	20	-8.00	0.713
Water Depth	BHS	-4.995	20	3.00	0.657
( <b>cm</b> )	SG	15.005	10	0.00	0.399
	М	15.005	20	8.00	0.704

### **2.4 Discussion**

Tree island plant communities that are complex and dynamic vary in spectral signatures, which are correlated with their composition, canopy density, standing biomass, and other characteristics. A suit of vegetation indices (VIs) that extract useful information based on the spectral signature of vegetation were evaluated in distinguishing tree island communities against the classifications based on *in situ* measurements of vegetation structural parameters. The resemblance among classifications was satisfactory (~80%), though it varied among individual islands and the regions of their occurrence. Our results suggest that certain VIs were suitable to distinguish different plant communities in and around tree islands, and thus can be used to detect vegetation changes over time.

In this study, twelve spectral vegetation indices (VIs) were evaluated for potential use in classifying vegetation in and around tree islands. However, not all indices were suitable in discriminating vegetation categories across all islands and regions. In general, the use of vegetation indices calculated as a ratio of normalized differences from only visible bands or from near infrared (NIR) and visible bands, especially the red band, is in common practice in studying different aspects of vegetation (Bannari et al. 1995). In our study as well, three of the five vegetation indices that performed best, i.e., MSR, RDVI and RDVI-LAI, were either simple ratio or normalized differences between red and NIR bands (Nagler et al. 2005; Haboudane et al. The red and NIR bands primarily contain more than 90% of vegetation related 2004). information (Bannari et al. 1995), and VIs based on these two bands commonly differentiate vegetation types based on their cover density i.e. biomass. Tree island vegetation communities, arranged along hydrologic gradient (Figure 2.7), also vary in productivity and biomass, from the low stature, unproductive marsh community to the highly productive hardwood hammock with the tallest trees (Ross and Jones 2004; Sah 2004). EVI and SLAVI, two other indices that were relatively efficient in discriminating target vegetation categories, also include both the red and NIR bands. However, they include an additional band as well. EVI uses the blue band, while SLAVI includes MIR. EVI has the ability to enhance the vegetation signature with improved sensitivity in high biomass region and a reduction in atmospheric influences (Nagler et al. 2005). SLAVI, designed for estimating specific leaf area (SLA), also accounts for MIR sensitivity to varying canopy structure in heterogeneous plant communities (Lymburner et al. 2000), thus making it efficient in differentiating target vegetation types in this study. In addition, SLAVI shows a strong relationship with wood volume in forests (Baruah et al. 2006). In tree islands, changes in tree basal area are common, as a result of succession, augmented by short or longterm below average water conditions, or disturbances like tropical storms and fire (Ruiz et al. 2013; see also Chapter 1 in this report). Thus, presence of SLAVI in composite indices used to classify tree island plant communities may be an advantage in monitoring vegetation change over time.

We were surprised to find that among the NIR and red band-based vegetation indices, NDVI performed poorly. NDVI is considered a surrogate measure of biomass and is widely used in vegetation studies (Rouse et al. 1973; Bannari et al. 1995; Jensen 2005), but did not do well in differentiating target vegetation categories in individual islands. In six of the nine islands studied, NDVI was not sufficiently discriminating among forest types or between sawgrass marsh and bayhead swamp. NDVI is known to become saturated in cases of dense multi-layered canopies, and also has a non-linear relationship with LAI. In Everglades tree islands, biomass increases along the hydrologic gradient, as does the depth of the canopy profiles that include varying degrees of understory vegetation, all contributing to an increase in LAI. The performance of the other two NIR and red band-based indices, DVI and IPVI, was also unsatisfactory. DVI is difference-based while IPVI is a ratio-based index, and both are considered computationally efficient versions of NDVI (Crippen 1990; Broge and Leblanc 2000). Though these indices have frequently been used in vegetation studies, they did not discriminate well among target vegetation categories in the tree islands. In contrast, improved indices like RDVI and MSR linearize their relationships with vegetation biophysical variables. RDVI combines the advantages of the DVI and NDVI, and MSR is a further improvement over RDVI (Haboudane et al. 2004), and thus both of these VIs were better choices to include in the group of indices used for classifying the vegetation into target groups.

In this study, four vegetation indices, GRVI, PPR, PVR, and WI performed very poorly in distinguishing vegetation types. Among them, GRVI, PPR and PVR use the green band, and the red or blue bands. Neither of them includes NIR. Other researchers have found that the spectral vegetation indices that use the green and red bands have a limited capability in retrieving vegetation information at full canopy cover (Vescovo & Gianelle 2008), a condition common in tree islands where canopy cover can be well above 80% (Sah 2004; Ruiz et al. 2011). However, the indices comprised of only the green band and the MIR band could be useful in differentiating vegetation types. The green band is sensitive to chlorophyll content, while the MIR can produce a good relationship with biomass (Everitt et al., 1989), and is sensitive to both water content and LAI; MIR reflectance is inversely related to leaf water content (Haboudane et al. 2004). In our study, this index was not included. Instead, MCARI1 uses green, red and NIR bands, and shows a strong relationship with LAI, and is considered useful for vegetation study (Haboudane et al. 2004). Its performance in our study was better than the four poorly performing indices discussed above.

The variation in the performance of vegetation indices in classifying tree island vegetation was not unusual, as the performance and suitability of a vegetation index are generally determined by the sensitivity of the index to a characteristic of interest. Beside the sensors and atmospheric elements, the spectral signature of vegetated areas also depends on a complex mixture of vegetation, soil color and brightness, soil moisture content, leaf distribution pattern in the canopy cover, leaf water content, among others (Bannari et al. 1995; Haboudane et al. 2004; Jensen 2005). In this study, target vegetation communities used to evaluate vegetation indices ranged between graminoid marshes and broadleaved sup-tropical hammocks, covering a wide range of hydrologic conditions (Figure 2.5), and both peat and mineral-rich soils (Jaychandran et al. 2004; Ross and Sah 2011). In addition, percentage of different lifeforms, canopy cover, and biophysical characteristics of leaf, including LAI, water content and nutrients also vary among these vegetation types (Sah 2004; Ross et al. 2006; Ruiz et al. 2011). Most importantly, the patchiness of vegetation and scale also play crucial role in performance of a vegetation index. In this study, we used Landsat TM bands for calculating vegetation indices. Landsat TM imagery has a pixel size of 30 m, which may cover a mixture of vegetation types as well as other ground elements in the same pixel, making an index less efficient in discriminating vegetation types. Moreover, the regional differences in tree island vegetation structure and composition might have also influenced the sensitivity of a vegetation index. Tree islands in ENP and WCAs differ in vegetation composition, even among hardwood hammocks (Ross et al. manuscript in preparation).

The scale of vegetation analysis also impacted the results of vegetation change over time. The analysis at the 30 x 30 m seem to be too coarse for the assessment of a change in vegetation type on the tree islands, as it did not capture the finer scale, decadal-scale variation recorded along the E-W transects or within plots (*see Chapter 1*). While within plot changes in woody abundance might not have resulted a change in vegetation types, broadly categorized in this study, the shift in species composition along the transects, primarily due to an increase in sawgrass cover, was also captured in VI-based analysis, especially in GL and SL, primarily because an increase in sawgrass cover over 10 years was noticeable in these islands. A finer scale imagery that has

more bands, and schematic classification trained over a larger data set covering an extensive, heterogeneous environment will improve the classification of sites.

The use of bi-seasonal (wet and dry season) atmospherically and geometrically corrected WV 2 (World Vision 2) data has been found more efficient for the detection of plant assemblages in Everglades marshes using the spectral signatures of vegetation (Gann et al. 2012). The WV 2 data have a spatial resolution of 2m a spectral resolution of 8 bands in the wavelength range of 400 - 1040 nm (WV 3: 8 additional bands in the 8 shortwave infrared wavelength range of 1195 – 2365 nm). Finally, the use of WV 2 for classifying vegetation in and around tree islands, followed by an evaluation of classification process using rigorous accuracy assessment processes is also recommended. An accurate classification of tree island vegetation can serve as a strong foundation for monitoring changes among tree vegetation types and between tree islands and marsh vegetation in response to hydrologic changes resulting from various restoration efforts that are underway as the part of CERP activities.

### Acknowledgments

We would like to acknowledge the assistance in field and lab (during the period between 2001 and 2012) provided by the following members of our lab: Dave L. Reed, Joshua Walters, Bernice Hwang, Darcy Stockman, Martha Elizabeth Struhar, David Jones, Susana Stoffella, Rafael Travieso, Erin J. Hanan, Mike Kline, Brooke Shamblin, Jessica Heinrich, Nate Colbert, Lawrence Lopez, Suresh Subedi, Cara Dodge, Danielle Crisostomo, Diana L. Rodriguez, Allison M. Lambert, Adam A. Spitzig and Jesus Blanco. We would also like to thank Everglades National Park Fire and Aviation Management Office for flight following and logistical support as well as HMC Helicopters. The project received financial support from the Department of Interior (Everglades National Park) and the RECOVER working group within the comprehensive Everglades Restoration Plan (CERP). The support from the RECOVER working group was provided through South Florida Water Management District (SFWMD), and U.S. Army Corps of Engineers (U.S. Army Engineer Research & Development Center). This study was allowed under ENP study EVER-00238.

#### **Literature Cited**

- Alexander, T.R. and A.G. Crook. 1984. Recent vegetational changes in South Florida. p. 199-210. In P.J. Gleason (ed.) Environments of South Florida: Present and Past II, second edition. Miami Geological Society, Coral Gables, FL, USA.
- Alexander, T.R., 1967, Effects of Hurricane Betsy on the southeastern Everglades: *The Quarterly Journal of the Florida Academy of Sciences* 39: 10-24.
- Allen, C.R., L.H. Gunderson and A.R. Johnson. 2005. The use of discontinuities and functional groups to assess relative resilience in complex systems. *Ecosystems* 8: 958-966.
- Armentano T.V., D.T. Jones, M.S. Ross, B.W. Gamble. 2002. Vegetation pattern and process in tree islands of the southern Everglades and adjacent areas. pp. 225-282 *In* F.H. Sklar and A. van der Valk (eds.) Tree Islands of the Everglades. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Armentano, T.V., J.P. Sah, M.S. Ross, D.T. Jones, H.C. Cooley, and C.S. Smith. 2006. Rapid responses of vegetation to hydrological changes in Taylor Slough, Everglades National Park, Florida, USA. *Hydrobiologia* 569: 293-309.
- Armentano, T.V., R.F. Doren, W.J. Platt and T. Mullins. 1995. Effects of Hurricane Andrew on coastal and interior forests of southern Florida: overview and synthesis. *Journal of Coastal Research* 21: 111-1444.
- Bannari, A., D. Morin, F. Bonn, A.R. Huete. 1995. A review of vegetation indices. *Remote Sensing Reviews* 13: 95-120.
- Baruah, P.J., T. Endo, T. Katsura, M. Setojima and Y. Yasuoka. 2006. Estimating timber-volume in a commercial *Eucalyptus globulus* plantation: results from two approaches. In: Proc. Asian Conference of Remote Sensing, October 9-13, 2006, Ulaanbaatar, Mongolia.
- Bazante, J., G. Jacobi, H.M. Solo-Gabriele, D. Reed, S. Mitchell-Bruker, D.L. Childers, L. Leonard and M.S. Ross. 2006. Hydrologic measurement and implications for tree island formation within Everglades National Park. *Journal of Hydrology* 329: 606-619.
- Bernhardt, C. 2011. Native Americans, regional drought and tree island evolution in the Florida Everglades. *Holocene* 2(1): 967-978.
- Bernhardt, C.E. and D.A. Willard. 2009. Response of the Everglades ridge and slough landscape to climate variability and 20<sup>th</sup> century water management. *Ecological Applications* 19: 1723-1738.
- Biggs, R., S.R. Carpenter, and W.A. Brock. 2009. Turning back from the brink: Detecting an impending regime shift in time to avert it. *Proceedings of the National Academy of Sciences of the United States of America* 106 (3): 826-831.
- Boughton, E.A., P.F. Quintana-Ascencio, E.S. Menges, and R.K. Boughton. 2006. Association of ecotones with relative elevation and fire in an upland Florida landscape. *Journal of Vegetation Science* 17: 361-368.
- Brandt, L.A., K.M. Portier, and W.M. Kitchens. 2000. Patterns of change in tree islands in Arthur R. Marshall Loxahatchee National Wildlife Refuge from 1950 to 1991. *Wetlands* 20: 1-14.
- Brock, J., C. McVoy, M. Oates, F. Santamaria, C.J. Saunders, T. Schall and E. Wunderlich. 2011. Landscape. In: F. Sklar, T. Dreschel and R. Stanek (eds). 2012 South Florida Environment Report – Chapter 6: Everglades Research and Evaluation.

- Broge, N.H. and E. Leblanc. 2000. Comparing prediction power and stability of broadband and hyperspectral vegetation indices for estimation of green leaf area index and canopy chlorophyll density. *Remote Sensing of Environment* 76: 156-172.
- Buma, B. and C.A. Wessman. 2011. Disturbance interactions can impact resilience mechanism of forests. *Ecosphere* 2:art64. doi:10.1890/ES11-00038.1
- Carr R.S. 2002. The Archaeology of Everglades Tree Islands. pp. 187-206. In F.H. Sklar and A. van der Valk (eds.) Tree islands of the Everglades. Kluwer Academic Publishers, Boston, Massachusetts.
- Chavez Jr., P.S. 1996. Image-based atmospheric corrections: Revisited and improved. *Photogrammetric Engineering and Remote Sensing* 62 (9):1025–1036.
- Cheng, Y., M. Stieglitz, G. Turk and V. Engel. 2011. Effects of anisotropy on pattern formation in wetland ecosystems. *Geophysical Research Letters* 38: L04402.
- Clark, M.W., M.J. Cohen, T.Z. Osborne, D. Watts, and T. Oh. 2009. Evaluating decomposition dynamics, community composition, and ridge-top senescence in the ridge-slough mosaic in response to climate change and water management. Annual Report 2009.
- Cornelius, J.M. and J.F. Reynolds. 1991. On determining the statistical significance of discontinuities within ordered ecological data. *Ecology* 72: 2057-2070.
- Crippen, R.E. 1990. Calculating the vegetation index faster. *Remote Sensing of Environment* 34: 71-73.
- D'Odorico, P., V. Engel, J. Carr, S.F. Oberbauer, M.S. Ross and J.P. Sah. 2011. Tree-grass coexistence in the Everglades freshwater system. *Ecosystems* 14: 298-310.
- Davidowitz, G. and M.L. Rosenzweig. 1998. The latitudinal gradient of species diversity among North American grashoppers within a single habitat: a test of the spatial heterogeneity hypothesis. *Journal of Biogeography* 25: 553-560.
- Espinar, J., M.S. Ross and J.P. Sah. 2011. Pattern of nutrient availability and plant community assemblage in Everglades tree islands, Florida USA. *Hydrobiologia* 667: 89-99.
- Everitt, J.H., D.E., Escobar and A.J. Richardson. 1989. Estimating grassland phytomass production with near-infrared and mid infrared spectral variables. *Remote Sensing of Environment* 30: 257–261, 1989.
- Ewe, S., J. Vega, K. Vaughan, and R. Bahe. 2009. Survey of living and ghost islands in Water Conservation Area 2A: Assessment of island microtopraphy, soil bulk density, and vegetation patterns, Volume 1. Final report to the South Florida Water Management District, West Palm Beach FL, USA. 24 November 2009
- Folke, C., S.R. Carpenter, B. Walker, M. Scheffer, T. Elmqvist, L.H. Gunderson and C.S. Holling. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology and Systematics* 35:557–581.
- Forys E.A. and C.R. Allen. 2002. Functional group change within and across scales following invasions and extinctions in the Everglades ecosystem. *Ecosystems* 5: 339-347.
- Furdi, M.A. and J.C. Volin. 2007. Tree island Hydrology and Ecology Project. Final Report Deliverable 8. Submitted to South Florida Water Management District, West Palm Beach, FL. 55 pp.
- Gann D, J.H. Richards and H. Biswas. 2012. Determine the effectiveness of vegetation classification using WorldView 2 satellite data for the Greater Everglades. Final Report submitted to South Florida Water Management District, West Palm Beach, FL. 62 pp
- Gawlik, D.E and D.A. Rocque. 1998. Avian communities in bayheads, willowheads, and sawgrass marshes of the central Everglades. *Wilson Bulletin* 110:45-55.

- Givnish, T.J., J.C. Volin, V.D. Owen, V.C. Volin, J.D. Muss and P.H. Glaser, 2008. Vegetation differentiation in the patterned landscape of the central Everglades: importance of local and landscape drivers. *Global Ecology and Biogeography*17: 384–402.
- Gosz, J.R. 1993. Ecotone hierarchies. *Ecological Applications* 3: 369-376.
- Gunderson, L. 1994. Vegetation of the Everglades: Determinants of Community composition. In: S.M. Davis and J.C. Ogden. Everglades: The Ecosystem and Its Restoration. Delray Beach, Florida.: St. Lucie Press: 323-340.
- Haboudane, D., J.R. Miller, E. Pattey, P.J. Zarco-Tejada and I.B. Strachan. 2004. Hyperspectral vegetation indices and novel algorithms for predicting green LAI of crop canopies: modeling and validation in the context of precision agriculture. *Remote Sensing ad Environment* 90: 337-352.
- Hagerthey, S.E., S. Newman, K. Rutchey, E.P. Smith and J. Godin. 2008. Multiple regime shifts in a subtropical peatlnad: community-specific thresholds to eutrophication. *Ecological Monographs* 78: 547-565.
- Hanan, E.J. and M.S. Ross. 2009. Across-scale patterning of plant soil-water interactions surrounding tree islands in Southern Everglades landscapes. *Landscape Ecology* 25:463-476.
- Hofmockel, K., C.J. Richardson, and P.N. Halpin. 2008. Effects of hydrologic management decisions on Everglades tree islands. C.J. Richardson (ed). pp. 191-214 *In* The Everglades Experiments: Lessons for Ecosystem Restoration. Springer, New York, NY
- Hofstetter, R.H. and F. Parsons. 1975. Effects of Fire in the Ecosystem: An Ecological Study of the Effects of Fire on the Wet Prairie, Sawgrass Glades and Pineland Communities of South Florida. Final Report Part 2. USDA National Park Service. EVER-N-48 NTIS No. BP 264463.
- Holling, C.S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4: 1-23.
- Jayachandran, K., S. Sah, J.P. Sah and M.S. Ross. 2004. Characterization, biogeochemistry, pore water chemistry, and other aspects of soils in tree islands of Shark Slough. pp. 29-40 In M.S. Ross & D.T. Jones (eds.), Tree islands in the Shark Slough landscape: interactions of vegetation, soils, and hydrology. Final report to Everglades National Park, Homestead, FL, USA, September 2004. Southeast Environmental Research Center, Miami, FL.
- Jensen, J.R. 2005. Introductory Image Processing: A Remote Sensing Perspective. 3<sup>rd</sup> Edition. Prentice Hall Series in Geographic Information System. Upper Saddle River, NJ. pp. 526.
- Jensen, J.R., K. Rutchey, M.S. Koch, and S. Narumalani. 1995. Inland wetland change detection in the Everglades Water Conservation Area 2A using a time series of normalized remotely sensed data. *Photogrammetric Engineering and Remote Sensing* 61(2):199-209.
- Johnson, L. 1958. A survey of the water resources of Everglades National Park, Florida. Report to Everglades National Park. <u>http://digitool.fcla.edu:80/R/-?func=dbin-jump-full&amp;object\_id=1018757&amp;silo\_library=GEN01</u>.
- Kline, M. M.S. Ross, P.L. Ruiz, B. Shamblin, J.P. Sah, E. Hanan and S. Stoffella. 2007. Marl Prairie/Slough Gradients; patterns and trends in Shark Slough and adjacent marl prairies. CERP monitoring activity 3.1.3.5. Third Annual Report. Dec 31, 2007. 26 pp.
- Knabb, R.D., J.R. Rhome and D.P. Brown. 2006. Tropical cyclone report Hurricane Katrina 23-30 August 2005. 43pp.

- Kolipinski, M.C. and A.L. Higer. 1969. Some aspects of the effects of quantity and quality of water on biological communities in Everglades National Park. Open File Report, USGS, Tallahassee, FL.
- Kumar, S., T.J. Stohlgern and G.W. Chong. 2006. Spatial heterogeneity influences native and nonnative species richness. *Ecology* 87:3186–3199
- Lago, M.E., F. Miralles-Wilhelm, M. Mahmoudi and V. Engel. 2010. Numerical modeling of the effects of water flow, sediment transport and vegetation growth of the spatiotemporal patterning of the ridge and slough landscape of the Everglades wetland. *Advances in Water Resources* 33: 1268-1278.
- Lemmon, P.E. 1956. A spherical densitometer for estimating forest overstory density. *Forest Science* 2: 314-320.
- Loope, L., M. Duever, A. Herndon, J. Snyder and D. Jansen. 1994. Hurricane impact on uplands and freshwater swamp forest. *BioScience* 44: 238-246.
- Loveless, C. M. 1959. A study of the vegetation in the Florida Everglades. *Ecology* 40:1-9.
- Ludwig, J.A. and J.M. Cornelius. 1987. Locating discontinuities along ecological gradients. *Ecology* 68: 448-450.
- Lymburner L., P.J. Beggs, and C.R. Jacobson. 2000. Estimation of canopy-average surface specific leaf area using Landsat TM data. *Photogrammetric Engineering and Remote Sensing* 66 (2): 183-191.
- MacArthur, R.H. and J.W. MacArthur. 1961. On bird species diversity. *Ecology* 42:594–598.
- Meshaka, W.E., R. Snow, O.L. Bass, and W.B. Robertson. 2002. Occurrence of wildlife on tree islands in the southern Everglades . pp. 391-427 In F.H. Sklar and A. van der Valk (eds.) Tree Islands of the Everglades. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Minchin P. 1998. DECODA: Database for Ecological Community Data. Anutech Pty. Ltd., Canberra, Australia.
- Motohka, T., K.N. Nasahara, H. Oguma, and S. Tsuchida. 2010. Applicability of green-red vegetation index for remote sensing of vegetation phenology. *Remote Sensing* 2: 2369 2387.
- Nagler, P.L., R.L. Scott, C. Westenburg, J.R. Cleverly, E.P. Glenn, and A.R. Huete. 2005. Evapotranspiration on western U.S. rivers estimated using the enhanced vegetation index from MODIS and data from eddy covariance and Bowen ratio flux towers. *Remote Sensing of Environments* 97: 337-351.
- Newman, S., J. Schuette, J.B. Gracce, K. Rutchey, T. Fontaine, K.R. Reddy and M. Peitrucha. 1998. Factors influencing cattail abundance in the northern Everglades. *Aquatic Botany* 60: 265 280.
- Olmsted, I. and T.V. Armentano 1997. Vegetation of Shark Slough, Everglades National Park. South Florida Natural Resources Center, Technical Report 97-001. National Park Service, Everglades National Park, Homestead, FL, USA.
- Palaseanu, M. and L. Pearlstine. 2008. Estimation of water surface elevations for the Everglades, Florida. *Computational Geosciences* 34:815–826.
- Pasch, R.J., E.S. Blake, H.D. Cobb III and D.P. Roberts. 2006. Tropical cyclone report Hurricane Wilma 15-25 October 2005. 27 pp.
- Patterson K. and R. Finck. 1999. Tree islands of the WCA 3A aerial photointerpretation and trend analysis project summary report. Report to the South Florida Water Management District. Geonex Corporation, St. Petersburg, FL.

- Pearlstine, L., S. Friedman, and M. Supernaw. 2011. Everglades Landscape Vegetation Succession Model (ELVeS) Ecolgoical and Design Document: Freshwater marsh and prairie component version 1.1. South Florida Natural Resource Center, Everglades National Park, Homestead, FL. 128 pp.
- Reed, D. and M.S. Ross. 2004. Hydrologic variation among and within tree islands of Shark Slough. pp. 5-16. In M.S. Ross and D.T. Jones (eds), Tree islands in the Shark Slough landscape: interactions of vegetation, soils, and hydrology. Final report to Everglades National Park, Homestead, FL, USA. September 2004.
- Risser, P.G. 1995. The status of the science examining ecotones. *BioScience* 45: 318–325.
- Ross, M.S. and D.T. Jones (Eds.). 2004. Tree islands in the Shark Slough landscape: interactions of vegetation, soils, and hydrology. Final report to Everglades National Park, Homestead, FL, USA. September 2004.
- Ross, M.S. and J.P. Sah. 2011. Forest resource islands in a sub-tropical marsh: soil-site relationships in Everglades hardwood hammocks. *Ecosystems* 14: 632-645.
- Ross, M.S., P.L. Ruiz, D.L. Reed,, K. Jayachandran, C.L. Coultas, J.P. Sah and M.T. Lewin, 2001. Assessment of Marsh Vegetation Responses to Hydrological Restoration in Shark Slough, Everglades National Park. Final Report (Cooperative Agreement 5280-9021). June 27, 2001. 102 pp.
- Ross, M.S., S. Mitchell-Brucker, J.P. Sah, S. Stothoff, P.L. Ruiz, D.L. Reed, K. Jayachandran and C.L. Coultas, 2006. Interaction of hydrology and nutrient limitation in the Ridge and Slough landscape of the southern Everglades. *Hydrobiologia* 569: 37–59.
- Rouse, J.W., R.W. Hass, J.A. Schell, and D.W. Deering. 1973. Monitoring vegetation systems in the Great Plains with ERTS. Proceedings of the Third ERTS Symposium, NASA SP-351, 1: 309-317.
- Ruiz, P.L., J.P. Sah, M.S. Ross, D. Rodriguez and A. Lambert. 2011. Monitoring of Tree Island Conditions in the Southern Everglades: The Effects of Hurricanes and Hydrology on the Status and Population Dynamics of Sixteen Tropical Hardwood Hammock Tree Islands. USACOE – US Army Engineer Research and Development Center. Cooperative Agreement: W912HZ-09-2-0019. 136 pp.
- Ruiz, P.L., J.P. Sah, M.S. Ross, and A.A. Spitzig. 2013. Tree island response to fire and flooding in the short-hydroperiod marl prairie grasslands of the Florida Everglades, USA. *Fire Ecology* 9 (1): 38-54.
- Ruiz, P.L., M.S. Ross and J.P. Sah. 2013. Monitoring of Tree Island Condition in the Southern Everglades: Hydrologic Driven Decadal Changes in Tree Island Woody Vegetation Structure and Composition. US Army Engineer Research and Development Center. Cooperative Agreement: W912HZ-09-2-0019. March 1, 2013. 41 pp.
- Sah, J.P. 2004. Vegetation structure and composition in relation to the hydrological and soil environments in tree islands of Shark Slough. p. 85-111. *In* M.S. Ross and D.T. Jones (eds), Tree islands in the Shark Slough landscape: interactions of vegetation, soils, and hydrology. Final report to Everglades National Park, Homestead, FL, USA. Sept. 2004.
- Sah, J.P., M.S. Ross, P.L. Ruiz, and S. Subedi. 2012. Monitoring of Tree Island Condition in the Southern Everglades. Annual Report-2011 submitted to US Army Engineer Research and Development Center. Cooperative Agreement #: W912HZ-09-2-0019. Modification # P00001. March 23, 2012. 72 pp.

- Sah, J.P., M.S. Ross, S. Saha, P. Minchin, and J. Sadle. 2014. Trajectories of vegetation response to water management in Taylor Slough, Everglades National Park, Florida. Wetlands 34 (Suppl 1): S65-S79.
- Saha, A.K., L.S. Sternberg, M.S. Ross and F. Miralles-Wilhelm. 2010. Water source utilization and foliar nutrient status differs between upland and flooded plant communities in wetland tree islands. *Wetlands Ecology and Management* 18: 343-355.
- Shamblin, B., M.S. Ross, S.F. Oberbauer, D. Gomez, L. Sternberg, A. Saha, and X. Wang, 2008. CERP monitoring and assessment program: tree island conditions in the southern Everglades. Annual Report for 2007 submitted to the South Florida Natural Resources Center, Everglades National Park, Homestead, FL.
- Sklar F., J. Richards, D. Gann, T. Dreschel, L. Larsen, S. Newman, C. Coronado Molina, T. Schall, C. Saunders, J. Harvey and F. Santamaria. 2013. Areal losses and gains in tree island habitat in Shark River Slough, Everglades National Park Inferences from 1952–2004 imagery analysis. In: 2013 South Florida Environmental Report (SFER), Vol. 1, Chapter 6. South Florida Water Management District, West Palm Beach, FL.
- Sklar, F.H. and A. van der Valk. 2002. Tree Islands of the Everglades: An overview. pp 1-18 *In*F.H. Sklar and A. van der Valk (eds.) Tree Islands of the Everglades. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Snyder, J.R., A. Herndon, and W.B. Robertson, Jr. 1990. South Florida rockland. pp. 230-277 *In* R. Myers and J. Ewel (eds.) Ecosystems of Florida. University of Central Florida Press, Orlando, Florida.
- StatSoft, Inc. 2006. STATISTICA (data analysis software system), version 7.1. www.statsoft.com.
- Stone, P.A. and G.L. Chmura 2004. Sediments, stratigraphy, and aspects of succession, chronology, and major prehistoric disturbance in the principarl type of large tree island in Shark Slough. pp. 45-82 *In* M.S. Ross & D.T. Jones (eds.), Tree islands in the Shark Slough landscape: interactions of vegetation, soils, and hydrology. Final report to Everglades National Park, Homestead, FL, USA, September 2004.
- Sullivan, P. 2011. Ground Water-Surface Water Interactions on Tree Islands in the Everglades, South Florida. A Ph.D. Dissertation. Florida International University, Miami, FL.
- Thomaz, S.M., L.M. Bini and R.L. Bozelli. 2007. Floods increase similarity among aquatic habitats in river-floodplain systems. *Hydrobiologia* 579: 1-13.
- Todd, M.J., R. Muneepeerakul, D. Pumo, S. Azaele, F. Miralles-Wilhelm, A. Rinaldo and I. Rodriguez-Iturbe. 2010. Hydrological drivers of wetland vegetation community distribution within Everglades National Park, Florida. Advances in Water Resources 33: 1279-1289.
- Troxler Gann, T., D.L. Childers and D.N. Rondeau. 2005. Ecosystem structure, nutrient dynamics, and hydrologic relationships in tree islands of the southern Everglades, Florida, USA. *Forest Ecology and Management* 214:11–27.
- USACOE 2014. Central Everglades Planning Project: Final Integrated Project Implementation Report and Environmental Impact Statement. US Army Corps of Engineers, Jacksonville District, FL.
- van der Maarel, E. 1990. Ecotones and ecoclines are different. *Journal of Vegetation Science* 1: 135-138.

- Vescovo, L. and D. Gianelle. Using the MIR bands in vegetation indices for the estimation of grassland biophysical parameters from satellite remote sensing in the Alps region of Trentino (Italy). *Advances in Space Research* 41: 1764-1772.
- Wade, D., J. Ewel and R. Hofstetter. 1980. Fire in South Florida ecosystems. U.S. Department of Agriculture and Forest Service., General Technical Report SE-17. 125 p. Southeast Forest Experimental Station. Asheville, N.C.
- Walker, S., J.B. Wilson, J.B. Steel, G.L. Rapson, B. Smith, W.M. King and Y.H. Cottam, 2003. Properties of ecotones: evidence from five ecotones objectively determined from a coastal vegetation gradient. *Journal of Vegetation Science* 14: 579-590.
- Warren, G. and G. Metternicht. 2005. Agricultural applications of high-resolutions digital multispectral imagery: evaluating within-field spatial variability of canola (*Brassica napus*) in Western Australia. *Photogrammetric Engineering & Remote Sensing* 71 (5): 595-602.
- Weiss, J.L., D.S. Gutzler, J.E. A. Coonrod, and C.N. Dahm. 2004. Long-term vegetation monitoring with NDVI in a diverse semi-arid setting, central New Mexico, USA. *Journal of Arid Environments* 58: 249-272.
- Wetzel, P.R. 2002. Analysis of tree island vegetation communities: hydrologic and fire impacts over a decade. p. 357-389. *In* F.H. Sklar and A. van der Valk (eds.) Tree Islands of the Everglades. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Wetzel, P.R., A.G. van der Valk, S. Newman, D.E. Gawlik, T. Troxler Gann, C.A. Coronado-Molina, D.L. Childers and F.H. Sklar, 2005. Maintaining tree islands in the Florida Everglades: nutrient redistribution is the key. *Frontiers in Ecology and the Environment* 3: 370–376.
- Wetzel, P.R., T. Pinion, D. Towles and L. Heisler. 2008. Landscape analysis of tree island head vegetation in Water Conservation Area 3, Florida Everglades. *Wetlands* 28: 276-289.
- Wetzel, P.R., A.G. van der Valk, S. Newman, C.A. Caronado, T.G. Troxler, D.L. Childers, W.H. Orem and F.H. Sklar. 2009. Heterogeneity of phosphorus distribution in a patterned landscape, the Florida Everglades. *Plant Ecology* 200: 83-90.
- Wetzel, P.R., F.H. Sklar, CA. Coronado, T.G. Troxler et al. 2011. Biogeochemical Processes on Tree Islands in the Greater Everglades: Initiating a New Paradigm. *Critical Reviews in Environmental Science and Technology* 41 (1) 670-701.
- Whittaker, R.H. 1960. Vegetation of the Siskiyou Mountains, Oregon and California. *Ecological Monographs* 30:279–338.
- Wiens, J.A., C.S. Crawford and J.R. Gosz, 1985. Boundary dynamics: a conceptual framework for studying landscape ecosystems. *Oikos* 45: 421-427
- Willard, D.A., C.E. Bernhardt, C.W. Holmes, B. Landacre and M. Marot. 2006. Response of Everglades tree islands to environmental change. *Ecological Monographs*, 76(4):565-583.
- Willard, D.A., C.W. Holmes, M.S. Korvela, D. Mason, J.B. Murray, W.H. Orem, and T. Towels. 2002. Paleoecological insights on fixed tree islands development in the Florida Everglades: I. environmental controls. pp. 117-151 *In* F.H. Sklar and A. van der Valk (eds.) Tree Islands of the Everglades. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Willoughby, H.L. 1898. Across the Everglades, a canoe journey of exploration. Fifth edition. Florida Classics Library Edition (1992), Port Salerno, Florida.

- Wilson, E.H. and S.A. Sader. 2002. Detection of forest harvest type using multiple dates of Landsat TM Imagery. *Remote Sensing of Environment* 80: 385-396.
- Wu, J., D. Wang, and M.E. Bauer. 2005. Image-based atmospheric correction of QuickBird imagery of Minnesota cropland. *Remote Sensing of Environment* 99: 315-325.
- Zaffke, M. 1983. Plant communities of water conservation area 3A; base-line documentation prior to the operation of S-339 and S-340. Technical Memorandum. South Florida Water Management District, West Palm Beach, FL, USA.
- Zweig, C.L., W.M. Kitchens. 2008. Effects of landscape gradients on wetland vegetation communities: Information for large-scale restoration. *Wetlands* 28: 1086-1096.





Figure 1.1: Conceptual model: vegetation dynamics in Shark River Slough tree islands and surrounding marsh.



**Figure 1.2:** Annual mean ( $\pm$  S.E.) and 30-Yr (1981-2010) average (dashed line) water level at the stage recorder P-33 located in Shark River Slough within Everglades National Park.



**Figure 1.3:** Study area map showing the location of transects on three Shark River Slough tree islands.



**Figure 1.4:** Hydrograph (1991-2011) for the three stage recorders used to calculate hydroperiod and annual mean water depth for transect sites and plots on three tree islands..



**Figure 1.5:** Annual mean ( $\pm$  1.SE) and 21-year average (dashed line) water surface elevation at three stage recorders, P33, NP203 and G620 between 1991 and 2011.



**Figure 1.6:** Box-plots (Mean, 1.SE, and 95% CI) showing mean annual hydroperiod averaged over 7 years prior to sampling along hardwood hammock, bayhead forest, and bayhead swamp transects in three tree islands.



**Figure 1.7:** Canopy height, bedrock elevation, ground elevation, and normalized Bray-Curtis dissimilarity (Z-Score) based on species cover and life form cover along N-S transects on three Shark River Slough islands. In split-moving window analysis, the Z-scores were averaged over 4 window sizes (window with of 6, 8, 10, and 12). HH = Hardwood hammock, BH = Bayhead forest, BHS = Bayhead swamp.



**Figure 1.8:** Bi-plots of (a) site, and (b) species scores from two-dimensional non-metric multidimensional scaling (NMDS) ordination of species cover data collected at the sites along N-S transects on Shark Slough islands. Environmental vectors fitted in the ordination spaces represent the direction of their maximum correlation with ordination configuration.



**Figure 1.9:** Bedrock elevation, ground elevation, and normalized Bray-Curtis dissimilarity (Z-Score) based on species cover along W-E transects on three Shark River Slough islands. In split moving-window analysis, the Z-scores were averaged over 4 window sizes (window with of 6, 8, 10, and 12 sites) separately for 2001 and 2011 samples. HH = Hardwood hammock, BH = Bayhead, BHS = Bayhead swamp; M = Marsh.



**Figure 1.10:** Relationship between mean normalized Bray-Curtis dissimilarity (Z-Score) and absolute difference in elevation on three N-S transects, one each in Black Hammock, Gumbo Limbo and Satinleaf tree islands.



**Figure 1.11:** Box plots showing the mean ( $\pm$ SE) of Beta diversity ( $\beta$ ) in 2001 and 2011 on the transects that crossed the head (hardwood hammock), middle (bayhead) and lower (bayhead swamp) portions of Shark Slough tree islands.



**Figure 1.12:** Relationship between mean Beta diversity ( $\beta$ ) and habitat heterogeneity (Coefficient of variation of Elevation) on nine transects in Shark Slough tree islands.



**Figure 1.13:** Bi-plots of site scores of centroids of sites, grouped by vegetation types, from twodimensional non-metric multidimensional scaling ordination of species cover data collected at the sites along W-E transects on Shark Slough islands. Environmental vectors fitted in the ordination spaces represent the direction of their maximum correlation with ordination configuration. WDep = Water depth (cm); HH = Hardwood hammock, BH = Bayhead forest, BHS = Bayhead swamp, M = Marsh.



**Figure 1.14:** Total graminoid cover in 2001 and 2011 and annual mean water level for seven years before sampling on the transects in three Shark Slough tree islands. Three transects are HH = hardwood hammock, BH = Bayhead, and BHS = Bayhead Swamp.



**Figure 1.15:** Cattail (*Typha domingensis*) cover in 2001 and 2011 on the bayhead swamp transect of Gumbo Limbo tree island.



**Figure 1.16:** Tree density (stems ha<sup>-1</sup>) and basal area (m<sup>2</sup> ha<sup>-1</sup>) in hardwood hammock plot in three tree islands sampled in 2001/2002 and 2012. BL = Black Hammock, GL = Gumbo Limbo, SL = Satinleaf.



**Figure 1.17:** Tree and sapling density (stems ha<sup>-1</sup>) in (a) bayhead forest and (b) bayhead swamp plots in three tree islands sampled in 2001/2002 and 2012. BL = Black Hammock, GL = Gumbo Limbo, SL = Satinleaf.



**Figure 1.18:** Tree and sapling basal area ( $m^2 ha^{-1}$ ) in (a) bayhead forest and (b) bayhead swamp plots in three tree islands sampled in 2001/2002 and 2012. BL = Black Hammock, GL = Gumbo Limbo, SL = Satinleaf.



**Figure 1.19:** Tree and sapling importance value (IV) in (a) bayhead forest and (b) bayhead swamp plots in three tree islands sampled in 2001/2002 and 2012. BL = Black Hammock, GL = Gumbo Limbo, SL = Satinleaf.



Figure 1.20: Images of Gumbo Limbo in 1994 and 1999, showing the area of sawgrass die-off present in 1999.



**Figure 2.1:** Study area map showing location the nine islands sampled for vegetation along N-S transects between 2012 and 2014.


Spectral Indicies Vs. Dichotomous Key (9 Tree Islands)



**Figure 2.2:** Box-plots showing differences in mean vegetation index values among five vegetation types classified using the vegetation structure-based dichotomous key.



**Figure 2.3:** Sites on three tree islands (Black Hammock, Gumbo Limbo and Johnny Buck) located within Everglades National Park showing the vegetation types classified using expertdeveloped classification key, structural parameter-based cluster analysis, and spectral vegetation indices (VI)-based cluster analysis. The VIs were calculated from the 2011 Landsat TM imagery.



**Figure 2.4:** Sites on three tree islands (Satinleaf, Chekika and Heartleaf) located within Everglades National Park showing the vegetation types classified using expert-developed classification key, structural parameter-based cluster analysis, and spectral vegetation indices (VI)-based cluster analysis. The VIs were calculated from the 2011 Landsat TM imagery.



**Figure 2.5:** Sites on three tree islands (WCA3A-266, TI66 and WCA3B-12)) located within Water Conservation Areas 3A, 3B showing the vegetation types classified using expertdeveloped classification key, structural parameter-based cluster analysis, and spectral vegetation indices (VI)-based cluster analysis. The VIs were calculated from the 2011 Landsat TM imagery.



**Figure 2.6:** Sites on three tree islands (Black Hammock, Gumbo Limbo and Satinleaf) located within Everglades National Park showing the temporal pattern in vegetation types classified using spectral vegetation indices (VI)-based cluster analysis. The VIs were calculated from the Landsat TM imagery from three different years, 1985, 1998 and 2011.



**Figure 2.7**: Normalized Skewed Normal Distribution curves of hydroperiod (A) and mean annual water depth (B) for plant communities on tree islands (HH = Hardwood hammock; BH = Bayhead; BHS = Bayhead Swamp; SG = Sawgrass; M = Marsh)

## Appendices

			Trai	nsect	
Island	Transect/Plot	Begin	(West)	End	(East)
		Easting	Northing	Easting	Northing
	Main-Axis (N-S)	531349	2832694	530963	2832288
Black Hammock	Hammock (WE-1)	531247	2832665	531340	2832595
	Bayhead (WE-2)	531206	2832640	531312	2832557
	Bayhead swamp (WE-3)	530965	2832455	531123	2832323
	Main-Axis (N-S)	526089	2834954	525721	2834033
Cumbo Limbo	Hammock (WE-1)	525910	2834842	526127	2834760
	Bayhead (WE-2)	525828	2834772	526090	2834674
	Bayhead swamp (WE-3)	525422	2834227	525864	2834069
	Main-Axis (N-S)	524537	2838128	524319	2837619
Satinlaaf	Hammock (WE-1)	524432	2838038	524558	2838013
Satimear	Bayhead (WE-2)	524400	2837949	524505	2837922
	Bayhead swamp (WE-3)	524377	2837861	524481	2837815
			Plot (C	entroid)	
		Ea	sting	Nor	thing
	Hammock	531	291.3	2832	2644.9
Власк Наттоск	Bayhead	531	245.9	2832	2597.8
	Bayhead swamp	531	052.6	2832	2372.2
C	Hammock	526	021.7	2834	4815.7
Gumbo Limbo	Bayhead	525	985.8	2834	4723.6
	Bayhead swamp	525	740.7	2834	4101.3
	Hammock	524	491.8	2838	3035.2
Satinleaf	Bayhead	524	453.8	2837	7942.5
	Bayhead swamp	524	420.7	2837	7834.3

A.1.1: Coordinates (NAD1983, UTM, Zone 17) for transects and plots in three tree islands (Black Hammock, Gumbo Limbo, and Satinleaf) in Shark River Slough.

	r			2001			2011	
		Ν	$Mean \pm SD$	Minimum	Maximum	$Mean \pm SD$	Minimum	Maximum
Island	Transect				Hydroperiod	(days)		
Black Hammock	Hammock	24	$177 \pm 128$	0	365	$114 \pm 109$	0	346
	Bayhead	28	$247\pm79$	81	365	$160\pm92$	21	344
	Bayhead swamp	41	$310\pm 66$	26	365	$243\pm81$	1	357
Gumbo Limbo	Hammock	47	$245\pm150$	0	365	$210\pm140$	0	362
	Bayhead	57	$325\pm39$	231	365	$273\pm62$	142	358
	Bayhead swamp	94	$355\pm14$	313	365	$321\pm31$	246	359
Satinleaf	Hammock	28	$182\pm142$	0	357	$136\pm118$	0	324
	Bayhead	22	$292\pm61$	138	354	$225\pm67$	69	315
	Bayhead swamp	24	$319\pm29$	233	354	$254\pm33$	148	314
					Water depth	n (cm)		
Black Hammock	Hammock	24	$-11.8 \pm 38.4$	-91.0	33.3	$-24.3 \pm 38.4$	-103.5	20.8
	Bayhead	28	$9.5\pm11.8$	-11.8	33.0	$-3.0\pm11.8$	-24.3	20.5
	Bayhead swamp	41	$19.9 \pm 11.2$	-25.0	38.5	$7.4 \pm 11.2$	-37.5	26.0
Gumbo Limbo	Hammock	47	$10.1\pm36$	-60.7	61.8	$-2.4\pm35.9$	-73.1	49.3
	Bayhead	57	$27.3 \pm 10.4$	7.8	49.8	$14.8 \pm 10.4$	-4.7	37.4
	Bayhead swamp	94	$35.2\pm7.0$	22.2	51.0	$22.8\pm7.0$	9.7	38.5
Satinleaf	Hammock	28	$\textbf{-6.2} \pm \textbf{34.6}$	-64.6	37.7	$-20.0 \pm 34.6$	-78.4	23.9
	Bayhead	22	$19.4 \pm 10.7$	-7.9	34.5	$5.6\pm10.7$	-21.6	20.7
	Bayhead swamp	24	$23.6\pm4.7$	10.5	34.0	$9.9 \pm 4.7$	-3.2	20.3
Island	Plot				Hydroperiod	(days)		
Black Hammock	Hammock		0 ± 0	0	1	0 ± 0	0	0
	Bayhead		$180 \pm 15$	160	204	79 ± 18	56	108
	Bayhead swamp		254 ± 26	209	276	167 ± 30	116	196
Gumbo Limbo	Hammock		0 ± 1	0	2	0 ± 0	0	0
	Bayhead		198 ± 27	161	250	111 -28	77	167
	Bayhead swamp		327 ± 6	321	339	283 ± 8	277	302
Satinleaf	Hammock		0 ± 1	0	3	0 ± 0	0	0
	Bayhead		206 ± 27	150	247	113 ± 19	74	148
	Bayhead swamp		283 ± 28	232	312	205 ± 41	130	246
Black Hammock	Hammock		-78.1 ± 14.5	-94.1	-44.3	-90.1 ±14.5	-106.1	-56.3
	Bayhead		-2.0 ± 2.6	-5.4	2.6	-14.0 ± 2.6	-17.4	-9.4
	Bayhead swamp		10.3 ± 3.8	3.8	13.7	-1.7 ± 3.8	-8.2	1.7
Gumbo Limbo	Hammock		-55.7 ± 6.3	-65.4	-44.6	-66.9 ± 6.3	-76.6	-55.8
	Bayhead		$0.3.0 \pm 5.2$	-6.8	9.6	-10.9 ± 5.2	-18	-1.6
	Bayhead swamp		$26.4 \pm 1.4$	24.9	29.5	$15.1 \pm 1.4$	13.6	18.3
Satinleaf	Hammock		-58.7 ± 7.1	-67.8	-45.1	-74.0 ± 7.1	-83.1	-60.4
	Bayhead		4.2 ± 5.4	-6.7	12.1	-11.1 ±5.4	-22.0	-3.2
	Bayhead swamp		17.6 ± 4.4	9.4	21.9	2.3 ± 4.4	-5.9	6.6

A.1.2: Mean ( $\pm$  SD), Min and Max hydroperiod and water depth on the transects sampled in 2001 and 2011 in three tree islands. Mean was calculated over 7 years prior the sampling. In the HH, BH & BHS Plot, values were averaged of sup-plots within each plot.

**A.1.3:** Linear regression co-efficient, coefficient of variation  $(r^2)$ , and p-value for the relationship between hydrological parameters (7-year average hydroperiod and mean water depth) and normalized Bray-Curtis dissimilarity (Z-Score). N-S = North South, HH = Hardwood hammock, BH = Bayhead, BHS = Bayhead swamp; ns = not significant. N-S transects were not sampled in 2011.

			Hydro	operiod					
Tree island	Transact		20	001			20	)11	
	Transect	а	b	$r^2$	р	а	b	$r^2$	р
Black Hammock	N-S	-1.201	0.016	0.637	< 0.001				
	HH	-0.502	0.014	0.460	0.001	-0.063	0.012	0.361	0.007
	BH	-0.964	0.023	0.621	< 0.001	-1.483	0.023	0.68	< 0.001
	BHS	-1.191	0.043	0.626	< 0.001	-1.160	0.022	0.325	< 0.001
Gumbo Limbo	N-S	-1.747	0.019	0.471	< 0.001				
	HH	-0.714	0.009	0.222	0.002	-0.925	0.012	0.26	< 0.001
	BH	-0.618	0.019	0.073	0.052		1	ıs	
	BHS					-0.748	0.056	0.304	< 0.001
Satinleaf	N-S	-1.195	0.014	0.514	< 0.001				
	HH	-0.561	0.009	0.376	0.002	-1.013	0.014	0.454	< 0.001
	BH	-1.212	0.033	0.576	< 0.001	-1.002	0.036	0.659	< 0.001
	BHS		r	ıs			1	ıs	
			Water	r depth					
Black Hammock	N-S	-1.041	0.054	0.652	< 0.001				
	HH	0.346	0.021	0.225	0.040		1	ıs	
	BH	-1.052	0.156	0.663	< 0.001	-1.468	0.175	0.635	< 0.001
	BHS	-1.889	0.244	0.580	< 0.001	-1.095	0.154	0.25	0.002
Gumbo Limbo	N-S	-1.757	0.081	0.518	< 0.001				
	HH	-0.705	0.033	0.141	0.014	-1.002	0.048	0.284	< 0.001
	BH	-0.998	0.139	0.188	0.001		1	ıs	
	BHS					-1.061	0.365	0.364	< 0.001
Satinleaf	N-S	-1.012	0.050	0.451	< 0.001				
	HH	-0.296	0.026	0.223	0.023	-0.773	0.034	0.301	0.008
	BH	-1.154	0.189	0.526	< 0.001	-0.858	0.219	0.551	< 0.001
	BHS		I	ıs			I	ıs	

		Black Hammock							G	umbo	Lim	bo				Sati	nleaf		
Species	SPCODE	W	E1	W	E2	W	E3	W	E1	W	E2	W	E3	W	E1	W	E2	W	E3
		2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011
Acrostichum danaeifolium	ACRDAN	2.06	1.04	5.59	1.89	0.01	1.19	7.28	3.00	12.78	6.80	2.48	0.31			2.26	1.07		
Aeschynomene pratensis	AESPRA		0.10				0.30					0.02	0.22		0.19	0.02		0.02	0.10
Ampelopsis arborea	AMPARB	0.02	0.52					0.35	0.53	1.30	0.02				0.39				
Andropogan glomeratus	ANDGLO													0.54					
Annona glabra	ANNGLA	11.63	3.58	20.30	4.95	12.67	9.82	18.19	11.48	17.32	6.73	0.01	0.08	1.59	4.94	8.43	12.65	4.35	4.27
Annona glabra_seedling	ANNGLA-S	0.67	1.15	0.54	1.00	4.95	3.74	1.90	0.05	2.25	0.21	0.44		0.39	0.02	0.15	0.24	1.73	0.13
Apios americana	APIAME		6.33		13.23					0.04									
Aster carolinianus	ASTCAR				0.09		1.13	0.02			0.09	0.52	0.48				0.02		
Aster dumosus	ASTDUM			0.46															
Bacopa caroliniana	BACCAR	1.69	1.96	0.45	2.82	2.82	2.26	0.85	0.53	1.55	1.57		0.52	1.07		1.26	1.41	0.77	1.77
Baccharis halimifolia	BACHAL												0.05						
Bacopa monnieri	BACMON														0.46				0.94
Blechnum serrulatum	BLESER	3.27	1.65	2.38	3.95	5.45	7.00	5.11	5.51	9.26	9.72	9.88	7.61	3.20	4.94	8.20	11.11	3.65	13.29
Boehmeria cylindrica	BOECYL	0.17	2.58	0.48	0.71	0.04	0.55	0.16	0.07	1.17	0.02	0.22	0.10	0.21	0.19	0.07		0.02	
Bursera simaruba	BURSIM	12.17	3.50					12.38	7.37					4.41	3.48				
Bursera simaruba_seedlimng	BURSIM-S							0.05	0.01										
Caesalpinia bonduc	CAEBON								4.45										
Celtis laevigata	CELLAE	3.00	1.02					14.18	0.73					0.73	0.37				
Celtis laevigata_seedling	CELLAE-S	0.02	0.52					0.10	0.12										
Cephalanthus occidentalis	CEPOCC		0.94	0.11	1.41	0.82	14.69	1.50	0.48	2.40	4.29	3.63	9.69			4.07	1.52	14.48	1.15
Chrysobalanus icaco	CHRICA	20.83	27.71	23.02	27.41	0.30	4.00	13.15	11.61	2.08	2.56			15.23	15.17	5.76	14.65		
Chrysobalanus icaco_seedling	CHRICA-S	4.79	2.48	1.77	2.00	4.80		7.99	2.16	3.89	2.48		0.05	1.45	2.09	0.48	3.20		
Chrysophyllum oliviforme	CHROLI													11.80	9.83				
Chrysophyllum oliviforme_seedling	CHROLI-S													0.09	0.19				
Cissus verticillata	CISVER	0.94	0.94	0.13	0.54			0.47	0.89	0.04	1.47			5.86	2.28				
Cladium mariscus ssp. jamaicense	CLAJAM	8.65	19.40	7.88	16.71	20.80	35.89	12.65	12.10	10.54	20.87	5.51	53.73	11.20	14.41	17.09	43.98	10.10	53.08
Coccoloba diversifolia	COCDIV														3.11				
Coccoloba diversifolia_seedling	COCDIV-S													0.36	0.11				
Cyperus haspan	CYPHAS					0.24	0.71	0.06	0.05	0.06		0.21	0.06						0.52

A.1.4: Mean species cover on the transects in three Shark Slough tree islands sampled in 2001/2002 and 2011. Three transects are: HH = Hammock, BH = Bayhead, and BHS = Bayhead Swamp. Tree species in the seedling (height <1 m) layer are listed separately.

		Black Hammock							G	ımbo	Lim	bo				Sati	ıleaf		
Species	SPCODE	W	E1	W	E2	W	E3	W	E1	W	E2	W	E3	W	E1	W	E <b>2</b>	W	E3
		(H	H)	( <b>B</b>	H)	(BI	HS)	(H	H)	( <b>B</b>	H)	(B)	HS)	(H	H)	(B)	H)	(B)	HS)
		2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011
Cyperus ligularis	CYPLIG	0.00				0.00										0.04		0.04	
Cyperus odoratus		0.02				0.02								2.04	0.07	0.42	0.16		
Dalbergia ecastaphyllum	DALECA								0.05					2.84	0.37	0.43	0.46		
Dichanthelium commutatum	DICCOM								0.05									0.00	
Dichanthelium dichotomum	DICDIC																	0.02	
Diodia virginiana	DIOVIR			0.04				0.01	0.21	0.13		0.17		0.98		0.07		0.06	
Echinochloa crusgalli	ECHCRU					0.09						0.04							
Eleocharis caribaea	ELECAR																	0.02	
Eleocharis cellulosa	ELECEL	0.35	1.54	0.11	9.29	0.06	3.15	9.01	19.05	2.15	13.91	1.25	7.79	3.54	2.74	3.39	13.65	9.40	22.92
Eleocharis elongata	ELEELO												0.51						
Eleocharis interstincta	ELEINT										1.46		0.51						
Eugenia axillaris	EUGAXI	8.65	6.96					13.54	8.47					3.71	6.59				
Eugenia axillaris_seedling	EUGAXI-S	0.94	4.08					6.64	10.64					1.18	3.13				
Eupatorium leptophyllum	EUPLEP																		0.02
Ficus aurea	FICAUR				0.36			0.21	0.13		0.04			0.36					
Ficus aurea_seedling	FICAUR-S							0.21		0.01				0.02					
Fuirena breviseta	FUIBRE						2.25	0.09	0.32	0.12	0.76	0.11	1.68	0.02	0.37	0.02		0.19	0.44
Habenaria sp.	HABXXX						0.13												
Hibiscus grandiflorus	HIBGRA													0.04					
Hydrolea corymbosa	HYDCOR							0.52			0.05						0.11		0.42
Hymenocallis palmeri	HYMPAL											0.01	0.01						
Hyptis alata	HYPALA														0.09				0.10
Ilex cassine	ILECAS			0.45					0.21	2.93	1.26								
Ilex cassine_seedling	ILECAS-S			0.02		0.06	0.24		0.01	1.39	0.05								
Ipomoea alba	IPOALB									0.05									
Ipomoea sagittata	IPOSAG	0.02	0.31		0.09		0.55	0.07		0.12	0.03	0.02	0.50	0.02	0.74	0.04	0.22	0.04	0.63
<i>Ipomoea</i> sp.	IPOXXX				0.09				0.04		0.02				1.93				
Justicia angusta	JUSANG			0.02	0.04			0.04	0.03	0.03	0.01	0.24	0.84	0.02	1.93	0.04	0.43	1.67	0.54
Kosteletzkva virginica	KOSVIR											0.05	0.05						
Leersia hexandra	LEEHEX	0.02		0.18		0.65	0.19	0.36	0.06	0.01	0.38	0.01	0.01		0.91	0.02	2.24		1.65
Ludwigia alata	LUDALA			0.02		0.01	2.05					0.04	0.01		0.46	0.04	0.11		0.73
Ludwigia curtissii	LUDCUR													0.04					

	Black Hammock						G	umbo	Lim	bo				Sati	ıleaf				
Species	SPCODE	W	E1	W	E2	W	E3	W	E1	W	E2	W	E3	W	E1	W	E2	W	E3
_		(H	H)	( <b>B</b>	H)	(BI	IS)	(H	H)	( <b>B</b>	H)	(BI	IS)	(H	H)	( <b>B</b> )	H)	(BI	IS)
7 7	LUDDED	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011
Ludwigia repens	LUDREP		0.52	0.14	0.02	2.33	0.65	1.05	2.82	1.39	0.01		0.01	0.46	0.01			0.13	<b>5</b> 10
Magnolia virginiana	MAGVIR			0.02	0.71	1.04	0.65	2.06	1.10	3.22	2.73			0.09	0.91	4.43	7.41	2.06	7.10
Magnolia virginiana_seedling	MAGVIR-S				0.09	0.13	0.01	1.59		1.18	0.11					0.02	0.22	0.52	0.02
Melothria pendula	MELPEN	0.04				0.00	0.10	0.02		0.08	0.10			0.10		0.11	0.44	0.40	0.40
Mikania scandens	MIKSCA	0.06		0.05		0.33	0.13	0.01		0.18	0.18	0.23	1.57	0.13		0.11	0.11	0.19	0.42
Mitreola petiolata	MITPET											0.01		0.11					
Morella cerifera	MORCER	2.60	1.46	2.73	3.00	5.60	4.13		0.95	1.26	3.12		0.51	1.77	0.37	1.72	0.98	0.04	
Myrica cerifera_seedling	MORCER-S	0.06	0.02	0.02	0.16	0.32	0.35	0.05		0.46	0.05				0.56	0.04	0.22		
Myrsine floridana	MYRFLO							0.28	0.52					1.86					
Myrsine floridana_seedling	MYRFLO-S							0.06	0.68					0.02					
Nectandra coriacea_seedling	NECCOR-S								0.05										
Nephrolepis exaltata	NEPEXA		0.42		0.09														
Nymphoides aquatica	NYMAQU												0.05						
Nymphaea odorata	NYMODO											0.05	0.68						
Oeceoclades maculata	OECMAC		0.10						0.13		0.01								
Osmunda regalis	OSMREG										0.04								
Oxypolis filiformis	OXYFIL													0.02					
Panicum hemitomon	PANHEM		1.67	0.13	0.46	0.16	4.23	0.27	0.12		0.02	0.21	0.28	0.04	0.46			1.02	0.10
Panicum rigidulum	PANRIG					0.09		0.05								0.02			
Panicum virgatum	PANVIR						0.06												0.83
Parietaria floridana	PARFLO								0.06										
Parthenocissus quinquefolia	PARQUI	0.10	2.38	0.11	2.23	0.10	0.24	0.03	0.33					0.75	1.69				
Paspalidium geminatum	PASGEM		0.42		0.09	0.01						0.22	0.06	0.11					
Passiflora pallens	PASPAL														1.00				
Passiflora suberosa	PASSUB				0.02														
Passiflora sp.	PASXXX						0.06												
Peltandra virginica	PELVIR	1.17		0.73	0.36	11.39	1.79	0.12		0.05	0.09	1.52	0.36			0.04	0.22	0.08	
Persea borbonia	PERBOR	0.02				0.90	0.64	0.70		0.24									
Persea borbonia_seedling	PERBOR-S	0.02	0.21		0.09	0.06	0.62	0.01	0.02	0.61	0.05		0.01	0.07			0.02		
Pluchea rosea	PLUROS	0.02	1.04	0.02		0.09	2.42		0.06	0.01	0.05	0.01		0.02	0.09	0.26	0.04	0.17	0.21
Polygonum hydropiperoides	POLHYD	0.04	0.21	0.39	0.09	0.02	0.30	0.01	0.02	0.08	0.04	1.45	1.38			0.28			
Pontederia cordata	PONCOR	0.15	0.10	0.14	0.09	1.77	1.54	1.10	0.13	1.05	0.62	2.27	1.43	1.00		1.57	0.13	6.19	0.85

		Black Hammock							G	umbo	Lim	bo				Sati	nleaf		
Species	SPCODE	W	E1 H)	W	E2	W (B)	E3	W	E1 H)	W	E2	(BI	E3	W	E1	W	E2	W	E3
		2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011
Proserpinaca palustris	PROPAL		1.44		-	0.13	6.51	0.01	0.01	0.13	0.45	0.53	0.28		0.02		0.11	0.06	0.23
Psilotum nudum	PSINUD		0.10			0.01							0.05						
Pteridium aquilinum var. caudatum	PTECAU													0.38					
Rhynchospora colorata	RHYCOL								0.21										
Rhynchospora divergens	RHYDIV				0.88														
Rhynchospora inundata	RHYINU		0.42		1.23		1.83	0.29	1.27	0.49	1.11	0.33	0.05		0.09		1.41		0.52
Rhynchospora microcarpa	RHYMIC				0.88					0.01							0.11		
Rhynchospora miliacea	RHYMIL																0.02		
Rhynchospora tracyi	RHYTRA		0.10		0.09	0.07		0.02	0.11	0.01	0.04		0.21	0.11					
Rivina humilis	RIVHUM							3.78	0.06										
Sabal palmetto_seedling	SABPAL-S	0.04	0.02						0.01										
Sacciolepis striata	SACSTR	1.46		0.02		0.89		0.02		0.03		0.29							
Sagittaria lancifolia	SAGLAN	0.04	0.63		0.18	0.02	0.24				0.74		6.58	0.02			0.33		2.21
Salix caroliniana	SALCAR	5.40	4.50	12.38	10.16	0.84		3.47	1.57	10.97	7.18	7.73	10.65	3.02	4.67	6.33	0.43		0.63
Salix caroliniana_seedling	SALCAR-S	0.04	2.06		0.55	0.02		0.10		1.49		4.84	0.21				0.02		
Sarcostemma clausum	SARCLA	0.04		1.36	0.09	0.02	0.01	2.11	0.35	5.55	0.35	4.03	0.30	6.84		0.13	0.22		
Saururus cemuus	SAUCER	0.04	1.13	0.23	0.45	0.46	1.50	0.83	0.27	1.46	0.54			0.39	1.00	0.15		0.21	
Setaria magna	SETMAG											0.01							
Sideroxylon foetidissimum	SIDFOE	4.10	6.19					1.05	1.32										
Sideroxylon foetidissimum_seedling	SIDFOE-S	0.04	0.10						0.05										
Simarouba glauca	SIMGLA													0.02	0.09				
Simarouba glauca_seedling	SIMGLA-S														0.46				
Smilax bona-nox	SMIBON						0.12												
Smilax laurifolia	SMILAU					0.06													1.02
Solanum erianthum	SOLERI							0.01	0.79										
Thelypteris interrupta	THEINT	0.54	1.88	1.25	3.13			1.69	3.53	4.81	8.32			0.27	2.57	0.28	3.54		
Thelypteris kunthii	THEKUN							1.05	0.21		0.01								
Thelypteris palustris var. pubescens	THEPAL											0.27							
Tillandsia balbisiana	TILBAL										0.01								
Tilandsia fasiculata	TILFAS					0.01		0.10	1.85										
Tillandsia flexuosa	TILFLE							0.05		0.18	0.03								
Tillandsia paucifolia	TILPAU							0.06	0.05										

			Bla	ck H	amm	ock			G	umbo	Lim	bo				Sati	nleaf		
Species	SPCODE	W (H	E1 H)	(B)	E2 H)	(BI	E3 IS)	W (H	E1 H)	(B	E2 H)	W (Bl	E3 HS)	W (H	E1 H)	(B	E2 H)	W (Bl	E3 HS)
		2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011	2001	2011
Tillandsia recurvata	TILREC					0.01	0.06	0.51	0.05	0.04									
Tillandsia usneoides	TILUSN	0.04						0.01		0.04									
Tillandsia utriculata	TILUTR							0.01											
Typha domingensis	TYPDOM			0.02	0.45	0.15	0.24				0.04		5.42		0.37			0.02	1.04
Utricularia foliosa	UTRFOL	1.13		0.02	1.77	0.30	2.00	0.70	0.11	2.40	2.22	12.30	5.77		1.74				0.21
Utricularia purpurea	UTRPUR	0.10						0.02	1.04	1.31	6.81		4.30				0.43		0.52
Vallisneria americana	VALAME									0.05									
Vitis sp.	VITXXX	1.44	0.10		1.23		0.01	0.33											
Unkown sp1	XXX001				0.02		0.01												
Xyris sp.	XYR001						0.01		0.05		0.01								

**A.1.4:** A change in vegetation type at sites within the transition zone between vegetation assemblages on the transects in three Shark Slough islands. HH = hardwood hammock, BH = Bayhead, BHS = Bayhead swamp, M = Marsh. Species codes are according to Appendix A.2.3.

T-land	Tran	Mada	Vegetat	Vegetation type	
Island	sect	Meter	2001	2011	Major change in species: cover
Black Hammock	BW1	45	HH	BH	BURSIM & CELLAE decreased, SALCAR appeared
Black Hammock	BW1	75	HH	BH	BURSIM & EUGAXI not present in 2011
Black Hammock	BW2	10	BHS	М	ANNGLA decreased, CLAJAM, ELECEL 83%
Black Hammock	BW2	15	BHS	М	ANNGLA decreased, CLAJAM, ELECELincreased
Black Hammock	BW2	115	BH	BHS	ANNGLA, SALCAR decreased, CLAJAM increased
Black Hammock	BW2	120	BHS	М	ANNGLA decreased, CIADIUM increased
Black Hammock	BW3	185	BHS	М	ANNGLA decreased, CLAJAM 83%
Gumbo Limbo	GW1	30	М	BHS	ANNGLA & CEPOCC increased, ELECEL also increased
Gumbo Limbo	GW1	150	HH	BH	BURSIM, FICUS absent
Gumbo Limbo	GW1	185	BHS	BH	Fern increased, MAGVIR increased
Gumbo Limbo	GW1	190	BHS	BH	MAGVIR present
Gumbo Limbo	GW1	195	BHS	М	MAGVIR decreased, CLAJAM & ELECEL increased
Gumbo Limbo	GW1	200	BHS	М	CEPOCC decreased, CLAJAM & ELECEL increased
Gumbo Limbo	GW1	210	BHS	М	ANNGLA decreased
Gumbo Limbo	GW1	220	BHS	М	ANNGLA decreased
Gumbo Limbo	GW2	40	M/BHS	М	CLAJAM increased
Gumbo Limbo	GW2	45	M/BHS	М	CLAJAM increased
Gumbo Limbo	GW3	170	BHS/M	М	CLAJAM increased
Gumbo Limbo	GW3	180	BHS	BHS/M	CLAJAM increased
Gumbo Limbo	GW3	190	BHS	BHS/M	CLAJAM increased
Gumbo Limbo	GW3	200	BHS	BHS/M	CLAJAM increased, CEPOCC decreased
Gumbo Limbo	GW3	420	М	BHS	SALCAR & CEPOCC increased
Satinleaf	SW1	100	BHS	BH	ANNGLA & CHRICA increased
Satinleaf	SW1	105	BHS	BH	ANNGLA & CHRICA increased
Satinleaf	SW1	110	BHS	BH	ANNGLA & CHRICA increased
Satinleaf	SW1	115	BHS	BH	CHRICA increased
Satinleaf	SW1	120	М	BHS	ANNGLA, SALCAR increased
Satinleaf	SW1	125	М	BHS	DALECA, MAGVIR increased
Satinleaf	SW2	25	М	BHS	ANNGLA increased
Satinleaf	SW3	20	BHS	BHS/M	High CLAJAM Cover - uniform
Satinleaf	SW3	25	BHS	BHS/M	High CLAJAM Cover - uniform
Satinleaf	SW3	30	BHS	BHS/M	High CLAJAM Cover - uniform
Satinleaf	SW3	90	М	BHS	CEPOCC decreased, CLAJAM & ELECEL increased
Satinleaf	SW3	95	М	BHS	MAGVIR increased

Tree Island	Plot	Easting_83	Northing_83	Plot	Easting_83	Northing_83
Black Hammock	1	531360	2832690	2	531330	2832660
Black Hammock	3	531300	2832630	4	531270	2832600
Black Hammock	5	531240	2832570	6	531210	2832540
Black Hammock	7	531180	2832510	8	531150	2832480
Black Hammock	9	531120	2832450	10	531090	2832420
Black Hammock	11	531060	2832390	12	531030	2832360
Black Hammock	13	531000	2832330	14	530970	2832300
Black Hammock	15	530940	2832270	16	530910	2832240
Black Hammock	17	530880	2832210	18	530850	2832180
Chekika Island	1	534420	2847690	2	534420	2847660
Chekika Island	3	534420	2847630	4	534390	2847600
Chekika Island	5	534390	2847570	6	534390	2847540
Chekika Island	7	534360	2847510	8	534360	2847480
Chekika Island	9	534360	2847450	10	534360	2847420
Chekika Island	11	534360	2847390	12	534330	2847360
Chekika Island	13	534330	2847330	14	534300	2847300
Chekika Island	15	534300	2847270	16	534300	2847240
Chekika Island	17	534270	2847210	18	534270	2847180
Chekika Island	19	534270	2847150	20	534240	2847120
Chekika Island	21	534240	2847090	22	534240	2847060
Chekika Island	23	534210	2847030	24	534210	2847000
Chekika Island	25	534210	2846970	26	534180	2846940
Chekika Island	27	534180	2846910	28	534180	2846880
Chekika Island	29	534150	2846850	30	534150	2846820
Chekika Island	31	534150	2846790	32	534120	2846760
Chekika Island	33	534120	2846730	34	534120	2846700
Chekika Island	35	534090	2846670	36	534090	2846640
Chekika Island	37	534090	2846610	38	534060	2846580
Chekika Island	39	534060	2846550	40	534060	2846520
Gumbo Limbo Hammock	1	526080	2834940	2	526080	2834910
Gumbo Limbo Hammock	3	526050	2834880	4	526050	2834850
Gumbo Limbo Hammock	5	526050	2834820	6	526020	2834820
Gumbo Limbo Hammock	7	526020	2834790	8	526020	2834760
Gumbo Limbo Hammock	9	525990	2834730	10	525990	2834700
Gumbo Limbo Hammock	11	525990	2834670	12	525960	2834670
Gumbo Limbo Hammock	13	525960	2834640	14	525960	2834610
Gumbo Limbo Hammock	15	525930	2834580	16	525930	2834550
Gumbo Limbo Hammock	17	525930	2834520	18	525900	2834520
Gumbo Limbo Hammock	19	525900	2834490	20	525900	2834460
Gumbo Limbo Hammock	21	525870	2834430	22	525870	2834400
Gumbo Limbo Hammock	23	525870	2834370	24	525840	2834370
Gumbo Limbo Hammock	25	525840	2834340	26	525840	2834310
Gumbo Limbo Hammock	27	525810	2834280	28	525810	2834250
Gumbo Limbo Hammock	29	525810	2834220	30	525780	2834220
Gumbo Limbo Hammock	31	525780	2834190	32	525780	2834160
Gumbo Limbo Hammock	33	525750	2834130	34	525750	2834100

**A.2.1:** Coordinates of the plots sampled along long-axis (N-S) transect on nine islands in Everglades National Park and Water Conservation 3A and 3B.

Tree Island	Plot	Easting_83	Northing_83	Plot	Easting_83	Northing_83
Gumbo Limbo Hammock	35	525720	2834070	36	525720	2834040
Gumbo Limbo Hammock	37	525690	2834010	38	525690	2833980
Gumbo Limbo Hammock	39	525660	2833950	40	525660	2833920
Gumbo Limbo Hammock	41	525630	2833890	42	525630	2833860
Heartleaf Hammock	1	547620	2848260	2	547620	2848230
Heartleaf Hammock	3	547620	2848200	4	547620	2848170
Heartleaf Hammock	5	547620	2848140	6	547620	2848110
Heartleaf Hammock	7	547620	2848080	8	547620	2848050
Heartleaf Hammock	9	547620	2848020	10	547620	2847990
Heartleaf Hammock	11	547620	2847960	12	547620	2847930
Heartleaf Hammock	13	547620	2847900	14	547620	2847870
Heartleaf Hammock	15	547590	2847840	16	547560	2847810
Heartleaf Hammock	17	547530	2847780	18	547500	2847750
Heartleaf Hammock	19	547470	2847720	20	547440	2847690
Heartleaf Hammock	21	547440	2847660	22	547410	2847630
Heartleaf Hammock	23	547380	2847600	24	547350	2847570
Heartleaf Hammock	25	547320	2847540	26	547290	2847510
Heartleaf Hammock	27	547260	2847480	28	547230	2847450
Johnny Buck	1	528300	2834790	2	528300	2834760
Johnny Buck	3	528270	2834730	4	528270	2834700
Johnny Buck	5	528240	2834670	6	528240	2834640
Johnny Buck	7	528210	2834610	8	528210	2834580
Johnny Buck	9	528180	2834550	10	528180	2834520
Johnny Buck	11	528150	2834490	12	528150	2834460
Johnny Buck	13	528120	2834430	14	528120	2834400
Johnny Buck	15	528090	2834370	16	528090	2834340
Johnny Buck	17	528060	2834310	18	528060	2834280
Johnny Buck	19	528030	2834250	20	528030	2834220
Johnny Buck	21	528000	2834190	22	528000	2834160
Johnny Buck	23	527970	2834130	24	527970	2834100
Johnny Buck	25	527940	2834070	26	527940	2834040
Johnny Buck	27	527910	2834010	28	527910	2833980
Johnny Buck	29	527880	2833950	30	527880	2833920
Johnny Buck	31	527850	2833890	32	527850	2833860
Johnny Buck	33	527820	2833830	34	527790	2833800
Johnny Buck	35	527760	2833770	36	527730	2833740
Johnny Buck	37	527700	2833710	38	527670	2833680
Johnny Buck	39	527640	2833650	40	527610	2833620
Johnny Buck	41	527580	2833590			
Satinleaf Hammock	1	524520	2838120	2	524520	2838090
Satinleaf Hammock	3	524490	2838060	4	524490	2838030
Satinleaf Hammock	5	524490	2838000	6	524460	2837970
Satinleaf Hammock	7	524460	2837940	8	524460	2837910
Satinleaf Hammock	, 9	524430	2837880	10	524430	2837850
Satinleaf Hammock	11	524400	2837820	12	524400	2837790
Satinleaf Hammock	13	524370	2837760	14	524370	2837730
Satinleaf Hammock	15	524370	2837700	16	524340	2837670
Satinleaf Hammock	17	524340	2837640	18	524310	2837610
Satinleaf Hammock	19	524310	2837580	20	524280	2837550

71028674907102867430740286737077028673107702867250
7102867430740286737077028673107702867250
740286737077028673107702867250
77028673107702867250
770 2867250
800 2867190
830 2867130
830 2867070
830 2867010
860 2866950
860 2866890
860 2866830
860 2866770
890 2866710
890 2866650
920 2866590
920 2866530
950 2866470
950 2866410
950 2866350
950 2866290
950 2866230
950 2866170
950 2866110
950 2866050
070 2853210
070 2853150
070 2853090
070 2853030
040 2852970
040 2852910
010 2852850
010 2852790
980 2852730
950 2852670
2857560
2857500
2857440
2857380
2857320
2857260
2857200
2857140
2857080
390 2857020
2856960
2856900
420 2856840

Tree Island	Plot	Easting_83	Northing_83	Plot	Easting_83	Northing_83
WCA3B-12	27	546420	2856810	28	546420	2856780
WCA3B-12	29	546420	2856750	30	546450	2856720
WCA3B-12	31	546450	2856690	32	546450	2856660
WCA3B-12	33	546450	2856630	34	546450	2856600
WCA3B-12	35	546480	2856570	36	546480	2856540
WCA3B-12	37	546480	2856510	38	546480	2856480
WCA3B-12	39	546480	2856450	40	546480	2856420
WCA3B-12	41	546510	2856390	42	546510	2856360
WCA3B-12	43	546510	2856330	44	546510	2856300
WCA3B-12	45	546510	2856270	46	546540	2856240
WCA3B-12	47	546540	2856210	48	546540	2856180

**A.2.2:** Box-plots showing differences in mean vegetation index values (12 indices) among five vegetation types classified using the vegetation structure-based dichotomous key for nine tree islands.





IPVI Vs. Dichotomous Key (9 Individual Islands) • Mean 🗌 Mean±SE



MSR Vs. Dichotomous Key (9 Individual Islands) • Mean 🗌 Mean±SE



NDVI Vs. Dichotomous Key (9 Individual Islands)

• Mean 🗌 Mean±SE



PVR Vs. Dichotomous Key (9 Individual Islands)

• Mean 🗌 Mean±SE



RDVI-LAI Vs. Dichotomous Key (9 Individual Islands)

**A.2.3.** Matrices showing the comparison between plant community classifications achieved via the expert-developed classification key, structural data based cluster analysis, and spectral vegetation indices-based cluster analysis.

		Black Hammock (18)											
		Str	uctura	al Class	sifica	tion	S	pectra	al Classi	ificatio	on		
	CLASS	HH	BH	BHS	S	М	HH	BH	BHS	S	М		
sr	HH	1					0	1					
atic	BH		6	1				6	1				
ator	BHS			3	1				3	1			
ass	S				5					5			
öö	М				1	0					1		
۲ ۲	НН						0	1					
atio	вн							6					
fice	BHS								4				
tru Issi	S									6	1		
C a	M									U	Ó		
17780						Class	l sificati		mnariec	n	0/ Λ	aroomont	
						Dich	otomo		Structur	ni Al	70 A	as ss	
						Dich	otomo		Siluciul	ai I		00.00	
						DICH			specira			53.33 20.00	
						Struc	ctural v	's Spe	ctrai			58.89	
						la a 1 1			4			Ê.	
		0		(	Jum		nbo Isla	and (4	H)	e	2010		
1		Str	uctura	al Class	sifica	tion	5	pectra	al Classi	ficatio	on		
	CLASS	нн	BH	BHS	S	M	НН	BH	BHS	S	M		
sn uo	HH	2					0	2					
ati	BH		23	2				24	1				
sific	BHS			10				3	7				
chc ass	S			2	0				1	0	1		
σō	M					2					2		
Ę	HH						0	2					
atic	вн							23					
iți ctr	BHS							4	9		1		
stru ass	S									0			
0,0	м										2		
						Class	sificatio	on Co	mparisc	n	% A	areement	
						Dich	otomo	us vs s	Structur	al		90.24	
						Dich	otomo	IS VS	Spectra	I		80 49	
						Struc	ctural v	s Spe	ctral			82.93	
						onat	Julia	o ope	otrar		3	02.00	
					-	ohnny	Buck	(41)				6	
		Str	uctura		ifica	tion		nectra	I Classi	ficatio	n		
	CLASS	цц	RH		S	M		RH RH	BHS	S	M		
		1		DHO	0	IVI	0	1	DHO	0	IVI		
ous			10					0	4				
icat			10	10				9	1	4			
ioto ssifi	BHS			16	_				15	1			
lich	s			2	1				3	6			
	М			5		0			1	2	2		
- 5	НН						0	1					
ura	BH							9	1				
aifiic	BHS								16	5	2		
Stri ass	S								З	4			
Ū	М										0		
						Clas	sificatio	on Co	mparisc	n	% A	greement	
						Dich	otomo	us vs s	Structur	al		82.93	
						Dich	otomo	us vs s	Spectra	l l		78.05	
						Struc	ctural v	's Spe	ctral			70.73	

				1	Satir	nleaf H	lammo	ock (20	D)			]
		Str	uctura	al Class	sifica	tion	S	pectra	al Class	ificatio	n	
	CLASS	HH	BH	BHS	S	М	HH	BH	BHS	S	М	
sr	НН	1					0	1				
atic	BH		6					6				
affic at	BHS			9				1	8			
chc ass	S			2	0				2	0		
ΞÖ	М			2		0					2	
E	НН						0	1				1
ural atic	BH							6				
lific	BHS							1	10			
Stru ass	S									0		
ΰÖ	М										2	
						Clas	sificatio	on Co	mparisc	n	% A	greemen
						Dich	otomo	us vs s	Structur	al		80.00
						Dich	otomo	us vs s	Spectra	I		80.00
						Struc	ctural v	s Spe	ctral			90.00

					Ch	ekika	Island	d (40)		11.7		
	1.000	Str	uctura	al Class	sificat	ion	Spectral Classification					
	CLASS	HH	BH	BHS	S	М	НН	BH	BHS	S	М	
st	HH	2					0	2	1 · · · · · · · · · · · · · · · · · · ·			
atic	BH		16					16				
ific	BHS		1	0				1	0			
cho ass	S				8	4				12		
öö	M					9		1		5	3	
Ľ	HH						0	2				
atic	BH							17			(	
ific to	BHS								0			
Struass	S									8		
ΰÖ	M							1		9	3	
	•					Class	sificatio	on Co	mparisc	n	% A	

Agreement 87.50 Dichotomous vs Structural Dichotomous vs Spectral Structural vs Spectral 77.50 70.00

			Heartleaf Hammock (28)								
		Str	uctura	al Class	sifica	tion	S	on			
	CLASS	HH	BH	BHS	S	М	НН	BH	BHS	S	М
sr	HH	0					0				
atic	BH		14	1				13	2		
of or	BHS		4	3					7		
chc ass	S				6					6	
ΩÖ	M					0					0
Ľ	НН						0				
ural	BH							13	5		1
aifio	BHS								4		
Stru ass	S									6	
0	M										0
						Clas	sificatio	on Co	mparisc	on	% A
						Dich	otomo	us vs	Structur	al	
						Dich	otomo	us vs	Spectra	d l	

Structural vs Spectral 82.14

			WCA3A-266 (20)									
		Structural Classification					Spectral Classification					
	CLASS	HH	BH	BHS	S	М	HH	BH	BHS	S	М	
sr	HH	0					0		to bound be bound to			
atic	BH		9					9				
affic at	BHS			8				1	7			
chc ass	S			1	0				1	0		
ΞÖ	M			2		0					2	
Ę	НН						0	11731				
atic	BH							9				
lific uctu	BHS							1	8		2	
Stru ass	S									0		
ΰÖ	M										0	
	-					Class	sificatio	on Co	mparisc	n	% A	

Dichotomous vs Structural Dichotomous vs Spectral Structural vs Spectral

85.00 90.00 85.00

			TI-66 (39)												
		Str	uctura	al Class	sificat	ion	S	pectra	I Classi	ssification					
	CLASS	HH	BH	BHS	S	М	ΗН	BH	BHS	S	М				
st	HH	1					0	1		000000000000000000000000000000000000000					
atic	BH		11	2				4	9						
choton assifica	BHS		4	17				2	19						
	S			4	0				4	0					
öö	М					0					0				
Ľ	HH						0	1							
atic	BH							5	10						
ific	BHS							1	22						
Stru ass	S									0					
ΰö	М										0				

Classification Comparison% AgreementDichotomous vs Structural74.36Dichotomous vs Spectral58.97Structural vs Spectral69.23

			WCA3B-12 (35)											
		Str	uctura	al Class	sificat	ion	S	pectra	al Class	ificatio	on			
	CLASS	HH	BH	BHS	S	М	HH	BH	BHS	S	М			
st n	HH	2					2	10						
atic	BH		20					18	2					
ific	BHS			4				2	2					
cho ass	S				3	2			1	4				
ΰā	М					4					4			
Ę	HH						2							
atic	BH							18	2					
lific	BHS							2	2					
Stru ass	S								1	2				
ΰÖ	М									2	4			
						Class	sificatio	on Co	mpariso	n	% Ag	greemer		
			Dichotomous vs Structural											
						Dich	otomo	IC VC	Sportra	Ê	\$	85 71		

Dichotomous vs Spectral85.71Structural vs Spectral80.00